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National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Southeast Regional Office
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Issued March 27, 2020 F/SER31:KR
Revised July 30, 2020 SERO-2019-03111

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Dr. Jeffrey Reidenauer
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Office of Strategic Resources
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Sterling, Virginia 20166

Dear Drs. McCallister and Reidenauer:

The enclosed Biological Opinion ("Opinion") known as the 2020 South Atlantic Regional Biological Opinion for Dredging and Material Placement Activities in the Southeast United States (2020 SARBO) responds to your request for consultation with us, the National Marine Fisheries Service (NMFS), pursuant to Section 7 of the Endangered Species Act (ESA). The Opinion considers dredging and material placement activities under the jurisdiction of the United States Army Corps of Engineers (USACE) Civil Works and Regulatory Programs and dredging/sand mining in borrow sites in federal waters under the jurisdiction of the Bureau of Ocean Energy Management (BOEM) Marine Minerals Program in the Southeast United States from the North Carolina/Virginia Border through and including Key West, Florida and the Islands of Puerto Rico and the U.S. Virgin Islands. Activities considered under the 2019 SARBO include dredging (maintenance dredging, dredging/sand mining in borrow sites, and restoration dredging/muck dredging to improve water quality); dredge material placement (sand placement for beach nourishment, nearshore placement, placement in in ocean dredged material disposal site (ODMDS), upland placement, transportation of materials between dredging and material placement locations); geotechnical and geophysical (G&G) surveys, conducted by USACE,[1] necessary to complete dredging and material placement projects, and monitoring for and handling of ESA-listed species encountered during projects covered under this Opinion.

NMFS concludes that the proposed action is not likely to jeopardize the continued existence of ESA-listed species or result in adverse effects to designated critical habitats considered in the Opinion (Table 8). This Opinion includes an Incidental Take Statement (ITS), with associated Reasonable and Prudent Measures (RPMs) and Terms and Conditions (T&Cs). The RPMs and T&Cs incorporate elements of the proposed action that appropriately minimize the impact of incidental take. Because NMFS has concluded the Project Design Criteria (PDC) include the measures necessary and appropriate to minimize the

[1] BOEM has previously consulted on G&G activities in the Atlantic, including within the action area, under NER-2018-15093: Biological Opinion: Sand Survey Activities for BOEM's Marine Minerals Program: Atlantic and Gulf of Mexico.

impact of incidental take of incidental take, the RPMs/T&Cs impose no additional requirements beyond those specified by the proposed action.

The requirements of this Opinion are separate and distinct from any requirement under other applicable laws, including the Marine Mammal Protection Act, the Magnuson-Stevens Fishery Conservation Act (MSA), and other federal, local, or state requirements. SARBO therefore does not replace consultation with the NMFS under the Essential Fish Habitat (EFH) provisions of the MSA. The routes of effect that NMFS evaluated under SARBO to determine if the actions proposed by USACE and BOEM are likely to jeopardize the continued existence of a listed species and/or result in the destruction or adverse modification of critical habitat differ considerably from the routes of effect NMFS evaluates during an EFH consultation to identify measures to avoid, minimize, or compensate for adverse impacts to EFH. Therefore, the PDCs in the 2020 SARBO should be viewed as neither substitutes for EFH conservation recommendations nor as necessarily sufficient steps for avoiding, minimizing, or compensating adverse impacts to EFH under the MSA.

We also note that NMFS and the United States Fish and Wildlife Service (USFWS) share jurisdiction over ESA-listed sea turtles. Therefore, USACE may need to consult with USFWS regarding impacts from beach nourishment activities, or any activity, that may affect sea turtles and/or their habitats in the terrestrial environment.

The project has been assigned a tracking number in our new NMFS Environmental Consultation Organizer (ECO), "SERO-2019-03111". Please refer to the ECO tracking number in any future inquiries regarding this project.

We look forward to further cooperation with you on other projects to ensure the conservation of our threatened and endangered marine species and designated critical habitat. If you have any questions on this consultation, please contact Karla Reece, Section 7 Team lead, by email at karla.reece@noaa.gov.

Sincerely,

Roy E. Crabtree, Ph.D.
Regional Administrator

Enclosure (s)
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Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	United States Army Corps of Engineers (USACE), and the Bureau of Ocean Energy Management (BOEM);
Activity:	South Atlantic Regional Biological Opinion for Dredging and Material Placement Activities in the Southeast United States (2020 SARBO) North Carolina/Virginia Border through and including Key West, Florida and the Islands of Puerto Rico and the U.S. Virgin Islands.
Consulting Agency:	National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office, Protected Resources Division, St. Petersburg, Florida Consultation Number SERO-2019-03111
Approved by:	____ Roy E. Crabtree, Ph.D., Regional Administrator NMFS, Southeast Regional Office St. Petersburg, Florida

Date Issued March 27, 2020
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Acronyms and Abbreviations

ATLO	Atlantic Ocean
BOEM	Bureau of Ocean Energy Management
CI	confidence interval
CITES	Convention on International Trade in Endangered Species of Wild Flora and Fauna
DDT	dichlorodiphenyltrichloroethane
DNA	deoxyribonucleic acid
DO	dissolved oxygen
DPS	distinct population segment
DTRU	Dry Tortugas
DWH	Deepwater Horizon oil spill
EF	essential feature
EPA	Environmental Protection Agency
EPR	eggs per recruit
ER	Engineering Regulation
ERDC	Engineer Research and Development Center
ESA	Endangered Species Act
EFH	Essential Fish Habitat
ETOF	equilibrium toe of fill
EWS	early warning system
FEMA	Federal Emergency Management Agency
G&G	geotechnical and geophysical
GIS	geographic information system
GOM	Gulf of Mexico
GPS	global positioning system
HRG	High-Resolution Geophysical
ITS	Incidental Take Statement
IWW	Intracoastal Waterway
JAXBO	Biological Opinion on the authorization of minor in-water activities throughout the geographic area of jurisdiction of the U.S. Army Corps of Engineers Jacksonville District, including Florida and the U.S. Caribbean (JAXBO)
MEC	Munitions of Explosive Concern
MMPA	Marine Mammal Protection Act
MSA	Magnuson-Stevens Fishery Conservation Act
NA DPS	North Atlantic DPS
NE	no effect
NLAA	not likely to adversely affect
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NCCOS	National Centers for Coastal Ocean Science
NP	not present
NWA DPS	Northwest Atlantic Distinct Population Segment
OCS	Outer Continental Shelf
ODESS	Operations and Dredging Endangered Species System
ODMDS	Ocean dredged material disposal site
Opinion	Biological Opinion
PBF	physical biological features
PCBs	polychlorinated biphenyls
PCE	primary constituent elements
PDC	project design criteria
PIT	passive integrated transponder

Plan	Risk Assessment and Risk Management Plan
PSO	protected species observer
Psu	practical salinity unit
RCP	Representative Concentration Pathway
RPM	Reasonable and Prudent Measures
S	seconds
SA DPS	South Atlantic DPS
SAC	USACE Charleston District
SAD	USACE South Atlantic Division
SAJ	USACE Jacksonville District
SARBA	SARBO Biological Assessment
SARBO	South Atlantic Regional Biological Opinion
SAS	USACE Savannah District
SAW	USACE Wilmington District
SMA	Southeast Seasonal Management Area
SSB/R	spawning stock biomass per recruit
T	threatened
TED	turtle excluder devices
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service
UXO	unexploded ordinance
WID	Water injection dredging

Units of Measurement

°C	degrees Celsius
°F	degrees Fahrenheit
cm	centimeter(s)
cm^2	square centimeter
CPUE	catch per unit effort
cy	cubic yards
dB	decibels referenced to 1 root mean square micropascal
ft	foot/feet
ft^2	square feet
kg	kilogram(s)
km	kilometer(s)
lb	pound(s)
m	meter(s)
m^2	square meter(s)
mcy	million cubic yards
mg/L	milligrams per liter
nmi	nautical mile(s)
PTS	permanent threshold shifts
RKM	river kilometer
RM	river mile
rms	root mean square
SPL	sound pressure level
TTS	temporary threshold shifts
μPa	micropascal

Introduction

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. §1531 *et seq*.), requires that each federal agency ensure that any action authorized, funded, or carried out by the agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of those species. When the action of a federal agency may affect a protected species or its critical habitat, that agency is required to consult with either the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the protected species or critical habitat that may be affected.

Consultations on most ESA-listed marine species and their designated critical habitat are conducted between the action agency and NMFS. Consultations are concluded after NMFS determines the action is not likely to adversely affect listed species or critical habitat or issues a Biological Opinion ("Opinion") that determines whether a proposed action is likely to jeopardize the continued existence of a federally listed species, or destroy or adversely modify federally designated critical habitat. The Opinion also states the amount or extent of listed species incidental take that may occur and develops nondiscretionary measures that the action agency must take to reduce the effects of said anticipated/authorized take. The Opinion may also recommend discretionary conservation measures. No destruction or adverse modification of critical habitat may be authorized. The issuance of an Opinion detailing NMFS's findings concludes ESA Section 7 consultation.

This document represents NMFS's Opinion based on our review of impacts associated with dredging (maintenance dredging, dredging/sand mining in borrow sites, and restoration dredging/muck dredging to improve water quality); dredge material placement (sand placement for beach nourishment, nearshore placement, placement in an ocean dredged material disposal site (ODMDS), upland placement, transportation of materials between dredging and material placement locations); geotechnical and geophysical (G&G) surveys necessary to complete dredging and material placement projects, and monitoring for and handling of ESA-listed species encountered during projects covered under this Opinion.

Activities will be authorized or permitted by the U.S. Army Corps of Engineers (USACE) Civil Works and Regulatory Programs in state waters and by the Bureau of Ocean Energy Management (BOEM) Marine Minerals Program in federal waters, in the Atlantic Ocean, from the North Carolina/ Virginia border south to Key West Florida and including the U.S. Virgin Islands and Puerto Rico.

We analyze the effects of these activities on the endangered (E) and threatened (T) species and designated critical habitat under our jurisdiction, in accordance with Section 7 of the ESA. This Opinion is based on information provided by USACE South Atlantic Division (SAD); USACE South Atlantic District Offices (Wilmington [SAW], Charleston [SAC], Savannah [SAS], Jacksonville [SAJ]); USACE Engineer Research and development Center, BOEM Marine Minerals Program; state partners from the North Carolina Wildlife Resource Commission, South Carolina Department of Natural Resources, Georgia, Florida Fish and Wildlife Conservation Commission, Florida Department of Environmental Protection; NMFS species and topic experts both within and outside of the NMFS Southeast Regional Office, and the best scientific and commercial data available.

Programmatic Consultations

NMFS relies on programmatic consultations as an effective tool to implement ESA Section 7(a)(2). "Programmatic consultation"[2] is defined as "a consultation addressing an agency's multiple actions on a program, region, or other basis."[3] Programmatic consultations allow for streamlined review of (1) multiple similar, frequently occurring, or routine actions expected to be implemented in particular geographic areas, or (2) Federal action agency programs, plans, policies, or regulations providing a framework for future proposed actions.[4] NMFS uses programmatic consultations to evaluate the effects of authorizing certain categories of frequently occurring activities or of agency policy or programs, where the specifics of any individual future project (either of the given category or type of activity, or occurring under the policy or program), such as the specific location, are not definitively known at the time of the programmatic consultation, but where there is a good understanding of the likely effects on resources listed under the Act. By consulting on the program, plan, policy, regulation, or suite of activities as a whole, NMFS is in a better position to comprehensively analyze the impacts of carrying out the programmatic action on ESA-listed resources.

As is done in this Opinion, a Programmatic Consultation generally identifies project design criteria (PDCs), which are the specific criteria, including the technical and engineering specifications, indicating how an individual project must be sited, constructed, or otherwise carried out both to be covered under this Opinion and to avoid or minimize adverse effects to ESA-listed species or designated critical habitat. The PDCs serve 2 important purposes. First, they ensure that the actions under consultation are sufficiently similar that their effects can be analyzed together. Second, the PDCs help protect species and critical habitat, and ensure that the action agency is meeting its obligation under Section 7(a)(2). In designing the PDCs, NMFS and the action agencies work to establish conditions that avoid or limit adverse effects on ESA-listed species and designated critical habitat. Where adverse effects cannot be avoided, the PDCs limit adverse effects to predictable levels that will not jeopardize the continued existence of listed species or destroy or adversely modify critical habitat, either at the individual project level or in aggregate. The 2020 SARBO includes PDCs that were developed during consultation with the action agencies and NMFS to include the measures that NMFS believes are necessary or appropriate to avoid or minimize impacts to ESA-listed species and designated critical habitat. The PDCs are considered part of the proposed action and must be followed in order for an activity to be covered under this Opinion.

The Programmatic Consultation evaluates the aggregate effects of categories of related actions or of the agency program. This includes the amount or extent of incidental take that is expected, if sufficient information exists to estimate take. Since programmatic consultations evaluate effects

[2] Note that the term "programmatic" is defined differently by NMFS in the context of a Programmatic Consultation or Programmatic Biological Opinion than it is by USACE when discussing a Programmatic *General Permit*.
[3] *See* 50 C.F.R. 402.02
[4] *See*, 50 C.F.R. 402.02; Joint Services memorandum, *Alternative Approaches for Streamlining Section 7 Consultation on Hazardous Fuels Treatment Projects*, https://www.fws.gov/endangered/esa-library/pdf/streamlining.pdf; 68 FR 1628 (January 13, 2003).

of expected future actions, the action agency must provide projections of the number of activities and the extent of expected effects from the proposed activities. The Programmatic Consultation must demonstrate that, when the PDCs are followed, the aggregate expected effect of all projects is not likely to adversely affect listed species or their critical habitat(s), or will not jeopardize ESA-listed species or destroy or adversely modify their critical habitat(s), as applicable. At the project-specific consultation stage, each proposed action is reviewed by the action agency or joint agencies to determine if it can be implemented according to the PDCs. For example, an action agency may certify that the expected effect of the project to be authorized is consistent with the PDCs and other conditions in the Programmatic Consultation. Adjustments to the project(s) may be necessary to bring the project(s) into compliance with the Programmatic Consultation document. Finally, the project-specific consultation procedures provide contingencies for proposed projects that cannot be implemented in accordance with the PDCs; for example, separate consultations may be performed on these projects if they are too dissimilar from those described in the Programmatic Consultation. In addition, the Programmatic Consultation provides a process for tracking the actual effects of the proposed activities, once implemented, to ensure that the number and scope of the effects do not exceed those analyzed in the consultation or otherwise require reinitiation. The process by which the SARBO Team, consisting of members of the USACE, BOEM, and NMFS, will conduct project and programmatic-level reviews of the actual effects and compliance with the Programmatic Consultation are defined by this Opinion (Section 2.9 of this Opinion).

The following elements, which generally are included in a Programmatic Consultation to ensure its compliance with ESA Section 7 and its implementing regulations, have been included in this Opinion:

1. Description of the proposed action (Section 2) and the PDCs (Appendix A-H) that are designed to avoid or minimize future adverse effects on listed species and critical habitat.
2. Procedures for streamlined project-specific review, and the process for separate consultations for projects that do not meet the requirements of the Opinion (Section 2.9).
3. Procedures for monitoring projects and validating the predicted effects, including the level of take, and for the comprehensive review of the projects authorized in reliance on the Opinion as a whole (Section 2.9).
4. Description of the manner in which the projects, when implemented consistent with the PDCs, may affect listed species and critical habitat and evaluation of expected effects of the covered projects (Sections 3 and Section 6).

The requirements of this Opinion are specific to the ESA-listed species and designated critical habitat units that occur within the action area and are subject to consultation with NMFS under Section 7 of the ESA. The requirements of this Opinion are separate and distinct from any requirement under other applicable law, including the Marine Mammal Protection Act, the Magnuson-Stevens Fishery Conservation Act, and other federal, local, or state requirements. Additional consultation may be required under the Marine Mammal Protection Act (MMPA), which requires authorization for take of marine mammals, and the Magnuson-Stevens Fishery Conservation and Management Act, which requires consultation whenever a federal agency authorizes, funds, or undertakes an activity that will affect essential fish habitat.

1 CONSULTATION HISTORY

There is a long history of dredging and material placement by the USACE in the southeast United States, and the USACE has consulted with NMFS on these dredging activities for nearly three decades. The first South Atlantic Regional Biological Opinion (SARBO) consultation on USACE dredging activities in the Southeast was completed in 1991(NMFS 1991), and subsequent Biological Opinions were completed in 1995 and 1997 (NMFS 1995; NMFS 1997). Previous SARBO consultations initially covered USACE dredging activities and then expanded to also include dredge material placement activities along the Atlantic coast from North Carolina through Florida. For simplicity, each new consultation or revision to SARBO will be referred to in this Opinion by the year it was completed (i.e., 1991 SARBO, 1995 SARBO, 1997 SARBO), and this new consultation will be referred to as the 2020 SARBO.

The USACE SAD requested to reinitiate consultation on the 1997 SARBO on April 30, 2007. The Minerals Management Service (reorganized and renamed as BOEM in 2010), was not party to this reinitiation request letter in 2007, but subsequently joined as a joint consulting agency with USACE SAD serving as the lead Agency. Similar to USACE, BOEM has a long history of engaging in dredging and material placement activities in the southeast.

On September 12, 2008, USACE SAD and BOEM provided a jointly prepared South Atlantic Regional Biological Assessment (SARBA) (USACE and USEPA 2008) to provide new information and analysis for a revised and updated SARBO. NMFS received the letter on September 15, 2008 and considers this date the official request for consultation. The conditions that led to reinitiation of consultation in 2007, completion of the 2020 SARBO in the form of a new Programmatic Biological Opinion, and the inclusion of activities in federal waters under the jurisdiction of BOEM in this Opinion, are described in detail in Appendix K. These decisions led to the USACE SAD and BOEM finalizing the PDCs for this Opinion and we initiated consultation on September 13, 2019.[5] On October 25, 2019, a draft of the Biological Opinion (NMFS tracking number SERO-2008-00000) was sent to the USACE and BOEM prior to final signature. Comments received on that draft resulted in additional negotiations completed on January 13, 2020.

1.1 Revision History

1.1.1 July 21, 2020: Appendix G, PDC GG.4 was revised to clarify the use of single beam sonar.

See Appendix L Revision History for a summary of the revision.

[5] Under the ESA consultation is initiated once NMFS has all information necessary to conclude the consultation. Because the PDCs that are the core of the project description were under development through the entire process, consultation was initiated very close to the issuance of this Opinion.

2 DESCRIPTION OF THE PROPOSED ACTION AND ACTION AREA

The proposed action for the 2020 SARBO is categorized into 5 types of activities, which are summarized below. Section 2.1 of this Opinion provides an estimate of the amount of work that will be completed annually under this Opinion, Section 2.2 discusses the authority under which the action agencies will complete the work, and Sections 2.3- Section 2.7 of this Opinion provide detailed descriptions of the proposed activities and the equipment used to complete these tasks. These activities will be carried out in waters from the North Carolina/Virginia Border through and including Key West, Florida and the Islands of Puerto Rico and the U.S. Virgin Islands, as further described as the action area in Section 2.8 of this Opinion.

1. Dredging
   - Maintenance dredging
   - Dredging/sand mining in borrow sites
   - Restoration dredging/muck dredging to improve water quality
2. Dredge material placement
   - Sand placement for beach nourishment
   - Nearshore placement
   - Placement in an ODMDS
   - Upland placement
3. Transportation of dredge materials between dredging and material placement locations, which is discussed by equipment type and as part of the dredge placement activities since the same equipment is used to transport and place the material.
4. G&G surveys performed by or authorized by the USACE necessary to complete dredging and material placement projects.[6]
5. Monitoring for and handling of ESA-listed species encountered during projects covered under this Opinion.

Use of this Opinion in Combination with other ESA Section 7 Consultations
Periodically, the USACE and/or BOEM may propose to authorize, fund, or carry out a dredging or nourishment project that is only partially covered by the 2020 SARBO. For example, activities that are consistent with the scale and scope of the 2020 SARBO, but involve dredging or disposal in areas not considered under the analysis for a particular activity, or construction of a new project, such as channel deepening or widening, that will subsequently be maintained in a manner consistent with the 2020 SARBO PDCs. These activities may be combined with this

[6] BOEM completed separate consultations with NMFS on G&G surveys they conduct as part of their marine minerals program, including those activities within the 2020 SARBO action area NMFS. 2019b. Sand survey activities for BOEM's Marine Minerals Program: Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, NER-2018-15093, Gloucester, MA.. As described in Appendix K (Section 2.5.4), NMFS determined that BOEM’s G&G activities, and related effects, are not caused by the proposed action.

Opinion, meaning the use of 1 Opinion (2020 SARBO) by reference to cover a portion of the action considered in a separate Section 7 consultation, and addressing the portions of the action not covered in 2020 SARBO within an additional consultation. Thus, if a project involves new construction (such as channel deepening or widening) in the action area, a separate Section 7 consultation would be completed for the construction of the project, and/or other project elements not considered in 2020 SARBO, while 2020 SARBO would be referenced to cover future maintenance. The additional Opinion would still consider the full scope of effects of the project at issue, to ensure that the entire project being implemented avoids jeopardizing ESA-listed species or destroying or adversely modifying critical habitat. While the additional Opinion must evaluate the full scope of effects associated with an action under separate Section 7 consultation, it would only authorize take not covered by the 2020 SARBO. This approach is intended to ensure that the maintenance activities and associated effects considered in this Opinion are considered and analyzed comprehensively and to avoid authorization of duplicate takes or impacts to ESA-managed resources.

NMFS, Southeast Regional Office, will differentiate between projects that are consistent with the nature and scope of the 2020 SARBO and may be combined in an additional consultation from those projects that are not appropriate to combine. Each consultation relying on 2020 SARBO to cover any portion of a proposed action will describe why it is appropriate to rely on 2020 SARBO.

Requirements of the Opinion (PDCs, Reasonable and Prudent Measures, or Terms and Conditions)
The PDCs (Appendices A-H) define the proposed action and provide the limitations of how, where, and when activities must be completed in order to be covered under this Opinion. The PDCs were developed by the SARBO Team as part of the proposed action, with the intent of avoiding or minimizing effects to ESA-listed species and designated critical habitat. All applicable PDCs will be incorporated into projects covered under this Opinion (e.g., as a special condition of a USACE Regulatory project permit or a contract condition for a USACE Civil Works or BOEM project).

This Opinion includes Reasonable and Prudent Measures (RPMs) and Terms and Conditions (T&Cs) (Sections 10.3 and 10.4). The RPMs and T&Cs incorporate those elements of the proposed action that NMFS has determined appropriately minimize the impact of incidental take. Because NMFS has concluded that the PDCs include the measures necessary and appropriate to minimize the impact of incidental take, the RPMs/T&Cs impose no additional requirements beyond those specified by the proposed action/PDCs.

2.1 Dredging and Placement Annual Estimates

As described in the Consultation History (Section 1 of this Opinion), both the USACE and BOEM have a long history of engaging in or authorizing dredging and dredge material placement within the action area. The USACE and BOEM provided information about previously completed projects and estimated the quantity of material to be dredged and placed annually under this Opinion to assist in quantifying details and effects of the projects anticipated to be covered under the 2020 SARBO. While it is anticipated that activities covered by this Opinion will continue to occur with the same frequency and volume as they have historically occurred, the ability to precisely quantify all project details was limited by (1) the nature of the reporting requirements in the 1997 SARBO, (2) differences between the proposed action considered in the 2020 SARBO and activities considered in previous SARBO consultations, including measures to minimize and avoid effects to ESA-listed species and designated critical habitat, and (3) the annual variation of both the location and quantity of projects completed as part of the proposed action. As explained in Section 2.9 of this Opinion, USACE/BOEM will provide an annual report of all projects under 2020 SARBO to NMFS as part of an annual review to confirm that the description of the proposed action remains accurate.

SARBA Appendix M Initial Estimate
The USACE and BOEM provided an initial estimate of the volume of dredging that will occur under the 2020 SARBO in the 2018 SARBA Appendix M. The estimate they provided was based on previous years of dredging (2014-2016) and anticipated dredging over a 5-year period, based on available data on past projects covered under the 1997 SARBO for routinely maintained Civil Works and larger Regulatory projects. It then considered the amount of work that at the time the USACE anticipated would be covered under the 2020 SARBO with the expansion of the Opinion from the 1997 SARBO, including projects authorized by USACE Regulatory, BOEM, and also projects that will occur in areas not considered under the 1997 SARBO (e.g., sturgeon rivers[7], Johnson's seagrass critical habitat, and the U.S. Caribbean). This initial estimate of the material to be dredged each fiscal year under the 2020 SARBO stated that USACE Civil Works dredges approximately 32 million cubic yards (mcy) of material each year in the SARBO action area. In addition, USACE Regulatory reported dredging approximately 16 mcy each year in the SARBO action area, acknowledging that this may be an overestimate, since USACE Regulatory beach nourishment projects may use materials from a previously reported USACE Civil Works project. The USACE estimated that half of the USACE Regulatory projects will include beach nourishment, resulting in approximately 8 mcy of beach material and nearshore placement each year in the SARBO action area.

USACE estimated in SARBA Appendix M that the maximum that industry could dredge in a year within the action area under this Opinion is approximately 58 mcy, which would include USACE Civil Works and USACE Regulatory projects, either of which might be BOEM-authorized projects, covered under this Opinion. Assuming that most years the dredging industry is only at 85% capacity (less than maximum due to windows, weather and other factor) that would mean up to approximately 49 mcy would be dredged in an average year. The initial estimate in SARBA Appendix M of 32 mcy for USACE Civil Works dredging plus 16 mcy for USACE Regulatory dredging equals 48 mcy per year, which aligns with the 49 mcy per year estimate of volume of dredging for an average year, with a maximum of 58 mcy per year. Since the customers for BOEM, in regards to sand resources, are the USACE or USACE permitted entities, SARBA Appendix M concluded that this estimate of dredging also encompasses BOEM's authorization for use of Outer Continental Shelf (OCS) sand resources.

USACE estimated that about 1/3 of the total amount dredged will be dredged using hopper dredges – approximately 16.3 mcy on average and 19.3 mcy maximum, based on the estimates provided in the SARBA Appendix M.

[7] For the purposes of this Opinion, sturgeon rivers are defined as the rivers that support Atlantic and/or shortnose sturgeon in the action area. These rivers are identified in the Sturgeon PDCs in Appendix E.

Updated Estimates
Since the completion of SARBA Appendix M in 2018, the USACE was able to compile additional information on the volume of dredging that occurred during each year since the 1997 SARBO within the 1997 SARBO action area (Table 1), which provided a more accurate account of past dredging than the initial estimate. The volumes reported under the 1997 SARBO were still generally limited to only USACE Civil Works projects. The USACE also provided updated information about the number of projects and volume of dredging estimated to occur in the first 5 years of implementation of the 2020 SARBO including updated estimated for projects in areas not previously covered under the 1997 SARBO such as those in sturgeon rivers, Johnson's seagrass critical habitat, or the U.S. Caribbean and a list of the volume of work routinely dredged for projects authorized by the USACE Regulatory. As in SARBA Appendix M, we still assume that the volume of material placed is expected to be approximately the same, but somewhat less than the volume dredged, since placement of material dredged under this Opinion may occur in upland disposal sites and on beaches above the mean high water line, meaning that only a portion of the material dredged will also be placed in areas under the jurisdiction of NMFS.

Table 1. USACE Civil Works Reported Dredging Totals under 1997 SARBO (1997-2018)

Year	Hopper CY	Total CY
1997	8,662,114	29,657,099
1998	5,657,819	24,866,920
1999	6,253,794	58,352,266
2000	14,821,757	28,036,368
2001	2,908,339	34,094,017
2002	9,065,303	56,295,417
2003	4,816,289	15,553,545
2004	4,836,651	15,573,907
2005	11,867,599	30,624,210
2006	6,875,942	28,270,826
2007	7,640,337	*35,653,012*[8]
2008	6,523,530	*35,653,012*
2009	14,382,100	65,015,600
2010	8,417,827	43,416,100
2011	6,987,091	*35,653,012*
2012	9,808,468	38,453,722
2013	7,362,809	31,838,422
2014	9,318,799	42,681,807
2015	7,120,000	40,814,000
2016	12,634,000	38,267,000
2017	10,417,000	28,752,000

[8] In years 2007, 2008, and 2011 (shown in italics), the total volume dredged was not reported, so the average volume of 35,653,012 cy dredged per year was added.

2018	9,102,000	26,844,000
Total CY (22 years)	185,479,568	784,366,262
Avg/Yr	8,430,889	35,653,012
Min/Yr	2,908,339	15,553,545
Max/Yr	14,821,757	65,015,600

We compared the information provided in Table 1 to the initial dredging volume estimate from the SARBA Appendix M and made the following observations:

- The average dredging volume reported per fiscal year based on a review of 5 years of dredging covered under the 1997 SARBO was 32 mcy, which is less than the 35.65 mcy reported annually under the 1997 SARBO when reviewing the data available for all years since it was completed in 1997 (Table 1).
- The maximum reported dredging in a single year since completion of the 1997 SARBO (Table 1) was 65.01 mcy, which is substantially higher than the SARBA Appendix M estimated maximum of 58 mcy of dredging that could be completed per year. In fact, 3 of the reported maximum dredging volume years in Table 1 exceeded the SARBA Appendix M estimated annual volume of 49 mcy.

Accordingly, the annual estimated dredge volumes used in this Opinion are based on the revised information of all reported dredging completed annually under the 1997 SARBO (Table 1), additional information provided in the 2018 SARBA Appendix M, additional information provided by the USACE regarding projects and estimated dredge volumes in the next 5 years and in areas not covered under the 1997 SARBO, and information on projects completed within the action area not previously covered under the 1997 SARBO that will be maintained in the future based on the expanded action area and PDCs. To complete the revised estimate, we considered the following dredging volumes:

1. Projects covered under the 1997 SARBO: We consider the total volume of USACE Civil Works dredging estimated annually using 1997 SARBO dredging volumes (Table 1) as the most accurate list of projects that will continue to be covered under the 2020 SARBO. These routine projects are typically associated with USACE Civil Work's projects, but also include some of the larger routine USACE Regulatory Projects.
2. USACE Regulatory Projects: We next considered the addition of USACE Regulatory projects not reported in Table 1; however, similar data available regarding dredging projects completed in the 1997 SARBO action area from 1997 – 2018 is unavailable. We accordingly rely on SARBA Appendix M to estimate that these projects will have a dredge volume of 16,000,000 cy. This estimate includes projects similar to the 1997 SARBO such as navigation dredging and beach nourishment, and also accounts for the additional estimated dredging associated with USACE Regulatory projects, including projects in secondary channels, ports, berths, and other areas not required to be maintained under Title 33, as well projects in areas outside of the 1997 SARBO action area.
3. Projects located in areas not covered under the 1997 SARBO: We next estimated the dredging volume for work in the U.S. Caribbean, which was not covered under the 1997 SARBO, and work in Atlantic sturgeon critical habitat and within the range of Johnson's seagrass, which was limited.

a. U.S. Caribbean: The USACE projects scheduled for the next 5 years that are expected to be covered under the 2020 SARBO include the 3 biggest and most common locations in the U.S. Caribbean that will require maintenance dredging (San Juan Harbor, Arecibo Harbor, and Mayaguez Harbor) of navigation channels. The projected dredging volumes for these locations and the dredging frequency are provided below in Table 2 and used to calculate the average dredging volume per location per year.

Table 2. New Dredging Areas

New Dredge Areas	Annual Dredge Volume (cy)	Frequency (years)	Annual Average
Mayaguez	1,000,000	10	100,000
Arecibo	1,000,000	10	100,000
San Juan	2,000,000	4	500,000
Total			700,000

b. Range of Johnson's seagrass: Johnson's seagrass were listed under the ESA in 1998, shortly after the completion of the 1997 SARBO, and work affecting Johnson's seagrass and its critical habitat was accordingly not covered under the 1997 SARBO. The projects in this region that would have been covered under the 1997 SARBO are limited to maintaining the Atlantic Intracoastal Waterway (referred to locally as the Intracoastal Waterway [ICW or IWW as referenced in this Opinion]), which was later covered under a USACE SAJ Regulatory Division regional general permit (SAJ-93 Maintenance Dredging of the Ports and Intracoastal Waterway within the range of Johnson's Seagrass, NMFS tracking number SER-2000-01199, signed June 4, 2001). The 2018 SARBA Appendix M estimated that 2.2 mcy of material will be dredged every 5 years within the range of Johnson's seagrass in the IWW (average = 444,000/year). Maintenance dredging in the IWW will account for the majority of projects within the range of Johnson's seagrass that will be covered under this Opinion, because most other maintenance projects in this area will be covered under another Programmatic Biological Opinion (*Biological Opinion on the authorization of minor in-water activities throughout the geographic area of jurisdiction of the U.S. Army Corps of Engineers Jacksonville District, including Florida and the U.S. Caribbean [JAXBO]*, NMFS Tracking Number SER-2015-17616)(NMFS 2017b).

c. Atlantic sturgeon critical habitat: Larger Civil Works and Regulatory projects in these rivers were covered under the 1997 SARBO prior to the designation of Atlantic sturgeon critical habitat in 2017, and USACE continued to complete projects in these areas under an ESA Section 7(a)(2)/7(d) analysis by the USACE while in reinitiation of consultation for this Opinion. Therefore, these larger projects are counted in the annual reported volumes in Table 1. Smaller projects, which may not have been covered by the 1997 SARBO, are accounted for in the SARBA Appendix M estimate of 16,000,000 cy. Therefore, no additional dredging volume was added to account for these projects.

d. ESA-listed corals: Projects within the range of ESA-listed corals were completed under the 1997 SARBO, until elkhorn and staghorn corals were listed and *Acropora* critical habitat was designated under the ESA in 2006. These projects are maintained infrequently and were completed under an ESA Section 7(a)(2)/7(d) analysis by the USACE while in reinitiation of consultation for this Opinion or an individual Section 7 consultation, and are counted in the annual reported volumes in Table 1. Therefore, no additional dredging volume was added to account for these projects.

Table 3. Estimated Combined Annual Hopper Dredging in the 2020 SARBO

	Minimum dredge volume per year (cy)	Maximum dredge volume per year (cy)	Average dredge volume per year (cy)
Reported USACE Civil Works annual dredging from Table 1	15,553,545	65,015,600	35,653,012
Estimated additional USACE Regulatory annual dredging for the 2020 SARBO (from SARBA Appendix M)	16,000,000	16,000,000	16,000,000
New Dredge Areas (USACE Civil Works in U.S.Caribbean)	700,000*	700,000*	700,000
New Dredge Areas (USACE Regulatory within range of Johnson's seagrass)	444,000*	444,000*	444,000
Total estimated volume dredged annually in the 2020 SARBO (Adds the 4 rows above)	32,697,545	82,159,600	52,797,012
Total anticipated hopper dredge estimated in the 2020 SARBO (assumes 1/3 of the dredge total is hopper dredging, per SARBA Appendix M)	10,899,182	27,386,533	17,599,004
**Where estimated minimum or maximum annual dredge volume is not available, average volume was used*			

To determine the total volume of annual dredging that is estimated to be completed by hopper dredge, the total estimated dredge volumes are combined in Table 3 and then divided by 1/3, based on SARBA Appendix M, which stated that 1/3 of all dredging is completed by hopper.[9] SARBA Appendix M estimated that 16.3 -19 mcy, with an average of 18.9 mcy, of material was expected to be dredged by hopper annually. *Our calculations, based on the updated estimated annual dredging totals in Table 3 estimate that hopper dredging covered under this Opinion will account for an annual dredge volume of between 10.90 and 27.39 mcy, with an average of 17.60 mcy dredged annually.* While the annual average estimated volume used for purposes of this

[9] The updated annual dredging totals provided by the USACE included a breakdown of the dredging method used for the reported USACE Civil Works projects. Only the USACE Civil Works projects reported from 1997-2006, provided a more comprehensive breakdown of the type of equipment used to complete the dredging. Using this data, hopper dredging made up 24% of all dredging, bucket dredging was 15%, cutterhead/pipeline was 51%, and other methods was 11%. Although this shows a general breakdown by dredging type for USACE Civil Works projects, these numbers do not account for the frequency of hopper dredging use in USACE Regulatory projects. Therefore, we believe it is still appropriate to use the USACE estimates that 1/3 of all dredging (Civil Works and Regulatory combined) is completed using hopper dredging. Using the 1/3 estimate is also a conservative approach for species conservation, in that hopper is expected to result in all take directly associated with dredging under the 2019 Opinion and therefore a worst case scenario when calculating future take.

Opinion (17.60 mcy) is similar to that provided in SARBA Appendix M (18.9 mcy), the maximum hopper dredging estimated volume is significantly higher than the original dredge volume estimates provided by the USACE and BOEM.

2.2 USACE and BOEM Delegation of Authority

The USACE and BOEM requested consultation under the joint consultation provisions of 50 CFR 402.07 for dredging and dredge material placement. Pursuant to 50 CFR 402.07, USACE is the lead agency for purposes of this Opinion. Both the USACE and BOEM can satisfy their ESA Section 7 consultation requirements by relying on this Opinion for projects meeting the requirements of this Opinion.

2.2.1 USACE

Two USACE Programs, the USACE Regulatory Program and the USACE Civil Works Program, have responsibility for authorizing and/or implementing the dredging and material placement activities evaluated in this Opinion. USACE Civil Works oversees navigation dredging, coastal storm risk management, and ecosystem restoration. USACE Civil Works projects are congressionally-authorized and federally-sponsored (i.e. federally-funded or partially federally-funded), meaning that the specific locations for dredging and material placement are congressionally-authorized and the projects are eligible for federal funding. For the purposes of this Opinion, projects that are managed by the USACE Civil Works Program are referred to as federally-**<u>authorized</u>**.

The USACE Regulatory Program permits dredging and material placement projects in accordance with Section 10 of the Rivers and Harbors act, Section 404 of the Clean Water Act, and Section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972. The specific location of USACE regulatory projects is not limited to congressionally-authorized locations and can include any in-water location under USACE statutory authority. However, regulatory projects may occur in areas where USACE Civil Program projects are congressionally-authorized, but not federally-funded, such as the maintenance of a channel or beach that is funded by a local municipality. For the purposes of this Opinion, projects managed by the USACE Regulatory Program are referred to as federally-**<u>permitted</u>**.

2.2.2 BOEM

BOEM is a Bureau within the Department of the Interior responsible for overseeing sand and gravel, oil and gas, alternative energy, and other mineral development on the OCS. The Outer Continental Shelf Lands Act of 1953 defines the OCS as submerged lands lying seaward of state's seaward boundary which, for states on the Atlantic is 3 nautical miles from the coast line (See 43 U.S.C. 1301, definition of "lands beneath navigable waters," and 43 USC Section 1331, definition of "outer Continental Shelf"). Thus, the use of minerals on OCS submerged lands, including the extraction of sand, is under the jurisdiction of BOEM. Under Public Law 103-426, if OCS sand resources are to be used for shore protection, beach restoration, or coastal wetlands restoration projects by Federal, State or local government agencies, or use in construction projects authorized by or funded in whole or in part by the Federal Government, BOEM may enter into a negotiated agreement that addresses potential use of OCS sand and gravel resources including with the USACE and other federal agencies. For purposes of this Opinion, BOEM is serving as a joint consulting agency, with the USACE serving as the lead agency for dredging/sand mining in federal waters.

2.2.3 Environmental Protection Agency (EPA)

While EPA has not requested consultation for the proposed action, EPA has jurisdiction over designation of ODMDS locations where dredging material may be placed under this Opinion. EPA and USACE have responsibility for ensuring that ocean dredged material disposal activities will not unreasonably degrade or endanger human health, welfare, amenities, or the marine environment under Sections 102 and 103 of the Marine Protection Research and Sanctuaries Act, as amended (33 U.S.C. 1412), also known as the Ocean Dumping Act. USACE SAD districts and EPA Region 4 work cooperatively in the management of the Ocean Dredged Material Disposal Program to ensure that each agency's responsibilities are met. Coordination occurs through formal review processes and informal staff communications. The specific coordination used by the 2 agencies at the time of the completion of this Opinion is outlined in the *Southeast Regional Implementation Manual for Requirements and Procedures for Evaluation of the Ocean Disposal of Dredged Material in Southeastern U.S. Atlantic and Gulf Coast Waters* (USACE and USEPA 2008). The purpose of this document is to provide guidance for applicants, permittees, and USACE SAD districts and EPA Region 4 staff evaluating ocean disposal of dredged material in southeastern U.S. coastal waters of the Atlantic Ocean and the Gulf of Mexico. The process will vary depending on whether the project is a USACE-sponsored Civil Works project or a project requiring a Marine Protection Research and Sanctuaries Act 103 permit. In those cases, where site designation by the EPA under Section 102 of the Marine Protection Research and Sanctuaries Act is required, the NEPA process applies and leads to the EPA issuing a rulemaking in the Federal Register establishing the site.

NMFS evaluated whether EPA's designation of ODMDS locations, where dredged material will be placed under the proposed action, met the definition for "effects of the action." "Effects of the action" are defined as effects "caused by the proposed action, including the effects of other activities that are caused by the proposed action. An effect or activity is caused by the proposed action if it would not occur but for the proposed action and is reasonably certain to occur." (50 C.F.R. § 402.02). Thus, NMFS' determination regarding whether an effect or activity is caused by the proposed action is governed by a "but for" standard of causation. Designated ODMD sites are used for disposal from a range of projects in the southeastern United States, including, but not limited to, projects covered by SARBO. Because these sites are utilized for a range of projects beyond those covered by SARBO, and therefore would continue to be designated and used regardless of the 2020 SARBO proposed action, NMFS does not consider EPA's designation of ODMDS to be a consequence of the 2020 SARBO proposed action.

2.3 Categories of Dredging

This section provides a general description of the categories of dredging activities covered under this Opinion, as specified in Appendices A-H. It also provides a brief description of the limitations to these forms of dredging based on the PDCs of this Opinion. Some of the forms of dredging and material placement discussed in Section 2.4 of this Opinion have multiple terms used to describe them, which are also explained with reference to other related forms of dredging or material placement. Information gathered to analyze the effects of placement covered under this Opinion is described in the additional consultation history information provided Section 2 of Appendix K. Specific techniques and equipment used for dredging are discussed in Section 2.5 of this Opinion.

2.3.1 Maintenance dredging

2.3.1.1 General description of maintenance dredging allowed under this Opinion

This proposed action includes maintenance dredging areas to the original federally-authorized or federally-permitted dredge template (depth, width, and total area). This includes maintenance dredging of navigation channels, including (1) federal waterways and channels required to be maintained under Title 33 (Navigation and Navigable Waters); (2) other navigation channels and canals (not required to be maintained under Title 33); and (3) other areas that have been previously federally-authorized or federally-permitted, already dredged or otherwise constructed, and need to be maintenance dredged under this Opinion to maintain the previously dredged template such as ports, berths, marinas, and areas around docks. The frequency with which an area needs to be maintained varies by the dynamic nature of the area and the need to maintain navigation. Therefore, a specific time interval/ dredging frequency cannot be specifically defined for each project.

Dredging of navigation channels may also incorporate advanced maintenance or channel realignment, as described below. The USACE has the authority to make minor modifications to the existing federally-authorized or federally-permitted dredge template of a project, as defined in Section 5 of the Rivers and Harbors Act of 1915 and Engineering Regulation (ER) 1165-2-119. This limited authority is executed under the Operations and Management program. These minor modifications are primarily used to improve the safety and efficiencies of existing ship traffic. All projects covered under this Opinion, are limited by the requirements in this Opinion, even if USACE authority to maintain channels is broader than the PDCs. Navigation improvements requiring new or amended Congressional authorization for construction are not covered under this Opinion and therefore would require an individual consultation with NMFS.

- <u>Advanced maintenance and overdepth dredging</u>: Dredging templates provide not only the federally-authorized or federally-permitted depth, but may also include an additional allowed dredging limit referred to as overdepth dredging and/or advanced maintenance. Overdepth dredging is the removal of additional amount of material to account for inaccuracies in the dredging process. Overdepth dredging is often defined in the original federally-authorized or federally-permitted dredging limit (e.g., a channel is authorized to -10 ft [feet] + 2 ft over depth). Advanced maintenance is dredging deeper and/or wider than the original dredge template in high shoaling areas that are expected to quickly fill, thereby reducing the dredging frequency in that portion of the navigational waterway.
- <u>Channel modifications, realignments, or bend easing</u>: The USACE has authority under its ER1165-2-119 to modify or realign the channel location. ER 1165-2-119 states:

> "Modification Under Existing Authority, Navigation Projects. The Chief of Engineers has but limited discretion with respect to modification of completed navigation projects without new authorization. The River and Harbor Act of 1909 provides (Section 6) an

authority for complete reconstruction of aged or outmoded lock and dam structures on authorized waterways and is permissive to modifications (in the replacements) to better serve navigation. This permits the USACE to study the need for such replacements with operations and maintenance funding; however, accomplishment of any recommended replacement project requires, as a minimum, the approval of the Secretary of the Army. Recommendations may, if they embody significant modifications, be submitted by the Secretary to Congress for specific authorization. The River and Harbor Act of 1915 provides (Section 5) an authority to increase channel dimensions, beyond those specified in project authorization documents, at entrances, bends, sidings and turning places as necessary to allow the free movement of vessels. Where not otherwise precluded by project authorization, the location of a completed channel may be altered during the course of the periodic maintenance program if the maintenance can thereby be more economically accomplished and related aids to navigation are readily adjustable to suit the restored channel dimensions at the shifted location."

2.3.1.2 Maintenance dredging covered under this Opinion

For the purposes of this Opinion, maintenance dredging is limited to the list of activities provided below that follow all relevant PDCs in this Opinion (see Appendices). The estimated volume of total dredging expected annually was provided and summarized in Section 2.1 of this Opinion. Equipment used for maintenance dredging is described in Section 2.5 of this Opinion.

- <u>Maintenance dredging in navigation channels (required to be maintained under Title 33):</u> Maintenance activities will occur at a frequency such that the area remains navigable, barring a sudden change from a storm, and that returning the area to the federally authorized or permitted dredge template does not alter the hydrology of the area. Maintenance dredging covered under this Opinion will be consistent with the PDCs of this Opinion, to limit effects to ESA-listed species or critical habitat to the effects considered in this Opinion. For example, dredging a channel that has not been maintained for a significant period of time and has thus returned to the surrounding conditions, is not considered maintenance.
    - Continued maintenance dredging of navigation channels required under Title 33 or provided in SARBA Appendix B, to the dredge template provided, including the defined overdepth and advanced maintenance depth. A summary list of these channels are provided in Section 2.8.1 of this Opinion.
    - Maintenance of navigation channels or canals where the deepening, widening, or new dredge area was analyzed in a separate ESA Section 7 consultation, dredged, and is maintained under this Opinion to the dredge template analyzed in the consultation including the defined overdepth and advanced maintenance depth.

- <u>Maintenance dredging in navigation channels and canals not required to be maintained under Title 33</u>: Maintenance dredging covered under this Opinion extends to dredging any channel or canal maintained for navigation within the action area (Section 2.8 of this Opinion) to the previously authorized or permitted dredge template such as those listed below. These include maintenance dredging in navigation channels such as the channels maintained for navigation that connect to main navigation channel, other smaller channels or canals maintained for navigation such as those in coastal communities and/or coastal neighborhoods, and maintained channels in rivers that are not part of the main navigation channel such as the secondary channel sections of a braided river. If another Programmatic Biological Opinion exists that covers this action, the other regional programmatic will be used, such as the Jacksonville Biological Opinion referred to as JAXBO, which is a programmatic that covers minor and maintenance dredging in Florida (NMFS 2017b).

- <u>Maintenance dredging areas other than navigation channels</u>: Maintenance dredging of an area to the previously permitted or authorized dredge template such as:
  - Ports and berths along a maintained navigation channels including those not owned and operated by a Port Authority or federally-required to be maintained such as in the individually maintained berths in Savannah Harbor.
  - Maintenance dredging in smaller areas such as public and private marinas, boat ramps, and around docks that were previously permitted by USACE and dredged.
- <u>Maintenance of sediment traps</u>: Some channels include a "sediment trap", or area dredged deeper and wider than the channel to collect sediment before it fills the navigation channel. For example, the sediment traps in Palm Beach and St. Lucie Inlet, Florida are designed to collect sand near the inlet that is then dredged/removed from the sediment trap location and used for beach placement. Maintenance dredging in SARBO includes continued maintenance of these existing sediment traps to the previous dredge template if completed in accordance with the PDCs in this Opinion.
- <u>Minor channel modifications, realignment, or bend easing</u>: Minor channel modifications considered under this Opinion are limited to minor realignments resulting from naturally shifting locations of the natural deep water location of the navigation channel or to address minor changes resulting from storm events (as defined in the General PDCs in Section 1of Appendix B). If intentional minor realignments (e.g. bend easings) are proposed, they will be considered under the Supersede procedures outlined in Section 2.9.5 of this Opinion.
  - <u>Minor channel modification following deep water</u>: An example of a location where minor channel modification or realignment may be necessary is a pass between 2 barrier islands. The deep water portion of the channel between the 2 islands may gradually shift over time based on the flow of water through the pass and the accretion of sand around an island. In this instance, shifting the navigation channel to align with the natural deep water area is the most cost effective and reasonable approach. The relocated navigation channel would maintain the same depth and width as the original channel and serve the same purpose as the originally permitted dredging of the channel.
  - <u>Minor channel modification to maintain an existing location</u>: If the natural shift of this channel strays too far in one direction, the USACE may decide to maintain the original location of the pass to accommodate the easiest navigational pattern from one location to another. This may require more dredging than following the current deep water location, but is still considered maintenance for the purposes of this Opinion. If the modification required is extensive, such as the closure of a pass by a hurricane, it would not be considered minor, but could be considered under the alternative review/SARBO Supersede procedures if the effects of the channel modification were considered substantially similar to the analysis in this Opinion (Section 2.9.5 of this Opinion).

- Dredge template modifications: The original dredge template of maintenance dredging areas (e.g., channels, ports, berths, marinas) may only be modified after the effects of the modification are analyzed in a separate ESA-Section 7 consultation or, if appropriate, through the SARBO Supersede process described in Section 2.9.5 of this Opinion. An example of a modification considered in a separate ESA Section 7 consultation is the deepening and widening of a port with all expansion of the original dredge template completed under the separate consultation and future maintenance of the deeper or wider dredge template covered under this Opinion.

2.3.2 Borrow Area Dredging (also referred to as Beach Sand Mining)

2.3.2.1 General description of Borrow Area Dredging

Borrow areas are in-water areas identified as containing beach quality sand that may be able to be used for beach nourishment projects. Borrow sites may be located in state waters, under the jurisdiction of the USACE, or federal waters, under the jurisdiction of BOEM. The selection of which borrow site to use is determined by either the USACE or the permittee based on a compatibility analysis of sediment quality, although other cost related factors are taken into consideration. Grain size, color, composition, and texture of the material are matched to the native sand as closely as practical to ensure proper project performance for projects such as beach nourishment. However, sand used for beach nourishment may also sometimes come from navigation channels as part of a maintenance dredging project or come from upland sources.

2.3.2.2 Borrow site dredging covered under this Opinion

For the purposes of this Opinion, borrow site dredging/sand mining dredging is limited to activities that follow all relevant PDCs in this Opinion. These include locations the USACE and BOEM provided in 2017 SARBA, which are summarized in Section 2.8.2 of this Opinion. In addition, new borrow sites within the action area may be covered under this Opinion if they meet the applicable PDCs of this Opinion including (but not limited to):

- Borrow area dredging is limited to a depth that does not result in hypoxic or anoxic conditions in the area (General PDCs in Section 1.1 in Appendix B). Hypoxic conditions are those with reduced dissolved oxygen and anoxic refers to areas with little to no remaining dissolved oxygen needed for most aquatic life to survive. Examples of dredging that may result in these conditions include the digging of step banked, deep holes that prevent water exchange. While this dredging practice was known to occur decades ago, this is not a current standard dredging method.
- Borrow sites that meet the General PDC requirements in Section 2.2 in Appendix B, for distance from hardbottom used by turtles for foraging or shelter.
- Borrow sites that meet the distance requirements in the PDCs that apply within the range of ESA-listed corals, for distance from coral hardbottom (Coral PDC Section 2 in Appendix C).
- Borrow sites that meet the PDC depth requirements that apply within the range of Johnson's Seagrass, for occurring in water depths not considered to support Johnson's seagrass(Johnson's Seagrass PDC Section 2.1 in Appendix D).

- Borrow sites that have undergone a separate individual Section 7 consultation (as contemplated in Section 2 of this Opinion).
- New borrow site locations that do not meet all of the PDCs in this Opinion, but are reviewed and approved through the SARBO Supersede procedures detailed in Section 2.9.5 of this Opinion.

2.3.3 Muck Dredging for Water Quality Enhancement

2.3.3.1 General description of Muck Dredging

Muck dredging is the removal of accumulated organic material typically found in areas with poor water quality. Equipment used for muck dredging can include hydraulic suction dredge, bucket dredge, or other similar dredging equipment. Muck dredging is used to improve the water quality or for restoration projects and is sometimes referred to as an environmental restoration project by the USACE. It is not intended to increase water depths to support vessel mooring or movement.

2.3.3.2 Muck dredging covered under this Opinion

For this Opinion, we define muck dredging as any dredging that involves the removal of muck sediments alone and does not remove the natural (non-muck) sediments. Muck dredging is covered under this Opinion if the project meets all of the applicable PDCs of this Opinion, including those specific to the types of dredging covered under this Opinion in Section 1 of Appendix B.

2.4 Categories of Dredge Material Placement

This section provides a general description of the categories of dredge material placement covered under this Opinion and in some instances the transportation method used for placement. It also provides a brief description of the limitations to these forms of placement based on the PDCs of this Opinion. Some of the forms of dredging and material placement have multiple terms used to describe it, which are also explained with reference to other related forms of dredging or material placement. The final dredge material placement site selected may vary depending on the proximity of the dredging operation to nearby available placement locations, costs associated with transporting the material, and the sediment dredged and its available uses (e.g., if it is beach quality sand it is typically used for a beach nourishment project). The equipment used to place material is described in Section 2.5 of this Opinion. All in-water placement activities are required to adhere to the PDCs including specific PDCs provided for in-water placement (Section 2.2 of Appendix B).

2.4.1 Sand Placement for Beach Nourishment

2.4.1.1 General description of sand placement for beach nourishment

Dredged material may be beneficially used (Section 2.4.3 of this Opinion) for beach nourishment projects if the navigation material is deemed to be beach quality sand or dredged from a designated borrow site (Section 2.3.2 of this Opinion). This material is placed on eroding beaches to enhance beach habitat for human and animal use and/or to provide storm risk management benefits to coastal structures. USACE participation in the restoration, protection, and placement of sediment on beaches is authorized under various statutes, including the Regulatory authority to permit beach nourishments under Section 10 of the Rivers and Harbors Act and Section 404 of the Clean Water Act. USACE coastal storm risk management projects reduce coastal erosion damages resulting from hurricanes and coastal storms, mostly through long-term beach nourishment projects, which currently involve intermittent placement of sand on shorelines for up to 50 years of federal participation under each Congressional authorization. Additionally, these projects provide for the continued use of these beaches for sea turtle and shorebird nesting that otherwise would be lost or compromised due to coastal erosion.

Sand is pumped to the beach by pipeline either directly from the dredge (e.g. cutterhead suction dredge) or from an offshore pumpout (e.g. hopper dredge) or hydraulic offload (e.g., scow) station and shaped using earth- moving equipment. The beach building process typically involves the use of bulldozers and other heavy equipment to distribute the sediment as it falls out of suspension at the outflow end of the pipeline. The sediment slurry is often diffused as it is released from the terminal pipe in order to reduce the flow velocity onto the beach and minimize the risk of creating scour holes. Dikes are typically constructed on 1 or 2 sides of the effluent area to allow for extended settlement time of suspended solids in order to reduce turbidity levels in the near shore environment. USACE reports it is unnecessary and impractical to artificially grade beach slopes below the mean low water elevation since they will be shaped by wave action to the natural slope. As such, the initial constructed profile extends seaward of the final adjusted design profile by a variable distance to support anticipated sand movement during and immediately after construction. Once sand distribution along the foreshore occurs, the adjusted profile is intended to resemble the planned design profile of the project.

The equilibrium toe of fill (ETOF) is defined by the USACE as the expected spatial extent of spreading of beach fill materials due to profile equilibration in the surf zone following construction events. In other words, it is the waterward extent that the sand placed is expected to extend once the project is complete. The ETOF is estimated using conservation of volume principles where an equilibrium profile, defined either by an analytical equation (Dean 1991) or by site-specific survey observations showing consistent profile shape (EM 1110-2-1100 Part V, Chapter 4 "Beach Fill Design") (USACE 2008), is translated seaward from the prefill profile until the construction template volume matches the equilibrium profile volume. Generally, this considers the natural slope of the beach sand at the shoreline prior to a beach nourishment event and then uses this profile to determine the extent the newly placed sand will extend once the natural slope returns (i.e., once it reaches equilibrium) after a placement event. Factors considered include the composition of the beach sand and the natural conditions of the area such as the currents, tides, and littoral drift; however, the USACE calculations to determine ETOF do not consider potential changes from storm events.

Placement of materials on the beach above the mean high water line is outside of the jurisdiction of NMFS and effects to ESA-listed species in these locations (e.g., sea turtles nesting) may require consultation with the USFWS.

2.4.1.2 Beach Nourishment covered under this Opinion

For the purposes of this Opinion, beach nourishment is limited to the list of activities provided below that follow all PDCs in this Opinion such as those designed to ensure placement of material does not obstruct species movement such as that of sea turtles entering or exiting the beach when nesting or species moving along the shoreline (General PDC Section 2.2 in Appendix B). Additional PDCs apply for any beach nourishment within the range of ESA-listed corals (Coral PDCs in Appendix C).

Beach nourishment covered under this Opinion:

- Beach nourishment in the locations and defined beach sand placement template described in Appendix B.
- Beach nourishment in areas that have undergone an individual Section 7 consultation and require a repeat nourishment event within the previously analyzed and filled beach sand placement template.
- Beach nourishment in areas reviewed through the SARBO Supersede procedures detailed in Section 2.9.5 of this Opinion.
- Placement on the uplands for activities with no intended equilibrium to occur in water (e.g., dune restoration) is outside of the jurisdiction of NMFS; however, any placement in the water and in-water activities related to beach placement (e.g., transport of materials through marine waters to the beach) are under the jurisdiction of NMFS and must adhere to the PDCs to be covered under this Opinion.
- New beach nourishment and placement is allowed outside the range of ESA-listed corals (defined in the Coral PDCs in Appendix C) if the new beach placement adheres to the PDCs in Section 1.2 of Appendix B. For the purposes of this Opinion, new beach placement is defined as placement of sand on an existing beach that has not previously been nourished.
- Beach sand placement outside of Florida will limit placement to be compatible with native sediment composition to minimize turbidity in the surrounding in-water environment. Most states have sand sediment composition requirements that address this concern.
- Outside of the range of ESA-listed corals, new beach placement is allowed if the design profile is similar/consistent to adjacent beaches. This does not include non-traditional beach nourishment designs such as those that protrude and may obstruct species movement along the shore.
- All new beaches (outside of the range of ESA-listed corals) are limited to placement in areas lacking hardbottom (e.g., worm-rock or other forms of non-coral hardbottom) and seagrasses that may be used as foraging or refuge habitat for ESA-listed species.

2.4.2 Nearshore Placement

2.4.2.1 General description of Nearshore Placement

Nearshore placement can include a number of different placement options with varying desired outcomes. Often nearshore placement refers to the placement of beach quality sand in the nearshore environment outside of the proposed beach template (i.e., outside of the ETOF) with the intent of feeding sand to the nearby beaches over time. Sand naturally moves along the shore within the littoral zone resulting in the redistribution of sand. Nearshore placement used for this purpose can be done by creating berms or mounds of sand either parallel to or perpendicular with the beach depending on the location and if the placement is also needed to dissipate wave energy by allowing the waves to break over the nearshore berm. Generally, these berms or mounds are a submerged formation.

Nearshore placement can also be used to deposit materials alongside or downdrift of a channel in an estuary, inland waterway, or river to allow placement closer to the dredged location and to keep the material in the system. This is usually performed with a side caster dredge and/or split hull hopper dredge in shallow draft navigation channels and inlets. Hopper dredges and cutterhead pipeline dredges can also perform this beneficial use. Material is pumped or placed downdrift of a navigation channel in a small mound where it is allowed to be transported by waves and currents. This approach provides environmental benefits by keeping sediment in the littoral system. The material placement can act as habitat improvement if placed in a manner that creates appropriate bottom relief to establish fish habitat, or placed at elevations suitable for the establishment of seagrass beds, oyster habitat, or other valuable ecosystem function.

2.4.2.2 Nearshore placement covered under this Opinion

For the purposes of this Opinion, nearshore placement is limited to the list of activities provided below that follow all PDCs in this Opinion. Additional PDCs within the range of Johnson's seagrass (Johnson's Seagrass PCDs, Appendix D), within the range of ESA-listed corals (Coral PDCs, Appendix C), and in sturgeon rivers (Sturgeon PDCs, Appendix E) are designed to protect against the burial of seagrasses and corals/coral hardbottom or the burial of sturgeon foraging resources in rivers where sturgeon occur by limiting where and when placement may occur in relationship to these resources.

Nearshore placement covered under this Opinion:

- Nearshore placement described in SARBA Appendix B (USACE 2017), which is generally related to beach nourishment projects.
- Nearshore placement in areas that have undergone an individual Section 7 consultation and require repeat placement within the previously analyzed and filled sand placement template
- Nearshore placement in areas reviewed and approved through the SARBO Supersede procedures detailed in Section 2.9.5 of this Opinion.
- New nearshore placement adjacent to beaches or through the use of side-casting material adjacent to a dredge location is allowed outside the range of Johnson's seagrass (Johnson's Seagrass PCDs, Appendix D), outside the range of ESA-listed corals (Coral PDCs, Appendix C), and outside of sturgeon rivers (Sturgeon PDCs, Appendix E) if it meets the following additional PDCs (General PDC Section 2.2 in Appendix B):
    - Placement does not occur in areas with hardbottom or other structural relief (e.g., worm-rock or other forms of non-coral hardbottom) or seagrasses that may be used as foraging or refuge habitat for ESA-listed species.

- Placement of material does not obstruct species movement such as that of sea turtles entering or exiting the beach when nesting, species moving along the shoreline, or through an area.
- Placement does not create a mound in loggerhead sea turtle critical habitat nearshore reproductive habitat that may result in structure that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures) or disrupt wave patterns necessary for orientation, and/or create excessive longshore currents.

2.4.3 Beneficial use

2.4.3.1 General description of Beneficial Use Placement

Beneficial use of dredged material is defined by USACE as "consistent with sound engineering practices and meets all federal environmental requirements, including those established under the Clean Water Act and the Marine Protection, Research, and Sanctuaries Act (see 33 CFR 335.7, 53 FR 14902)". The beneficial placement of material means that material dredged is able to be used for a desired purpose instead of a disposal site like an ODMDS.

The USACE considers beneficial use sites to include nearshore placement, placement alongside and downdrift of a navigation channel, and placement on a beach or other sandy habitat. Other beneficial uses include marsh creation, land creation, thin layer placement, fish and wildlife habitat enhancements, fisheries improvements, wetland restoration, etc.

2.4.3.2 Beneficial Use Covered Under This Opinion

This Opinion considers the beneficial placement of sand dredged during maintenance dredging for beach nourishment (described in Section 2.4.1) and disposal of material for nearshore placement (described in Section 2.4.2) if it meets the PDCs required for that form of placement activity. However, the USACE also considers other projects "beneficial use" if the material dredged can be used for activities like marsh creation, thin-layer placement (e.g., used for marsh creation or other disposal method), filling of holes to improve water quality, filling of holes or minor depressions to restore the appropriate depth for habitat restoration, or other similar placement activities. The effects from these other beneficial use activities are not considered under this Opinion; therefore an individual Section 7 consultation for such beneficial use would be required.

2.4.4 ODMDS

2.4.4.1 General description of an ODMDS

The EPA has the authority to promulgate ocean dumping criteria, designate recommended ocean disposal sites, and issue permits for dumping materials (except for dredged material) into ocean waters. Under Sections 102 and 103 of the Marine Protection Research and Sanctuaries Act, as amended (33 U.S.C. 1412), also known as the Ocean Dumping Act, the EPA and the USACE are responsible for ensuring that ocean dredged material disposal activities will not unreasonably degrade or endanger human health, welfare, amenities, or the marine environment. Marine

Protection Research and Sanctuaries Act Section 102 authorizes EPA to designate sites and times at which dumping may occur and to establish criteria for reviewing and evaluating permit applications. In those cases, where site designation by the EPA under Section 102 of the Marine Protection Research and Sanctuaries Act is required, the NEPA process applies and leads to the EPA issuing a rulemaking in the Federal Register establishing the ODMDS.

2.4.4.2 ODMDS Placement covered under this Opinion

The USACE and BOEM provided the locations of current ODMDS sites in SARBA and they are summarized in Section 2.8.3 of this Opinion. The proposed action for this Opinion limits the placement of material in ODMDS locations to designated ODMDS sites. Those designated at the time of completion of this Opinion are identified in Section 2.8. Placement at any ODMDS locations designated, or expanded, after the completion of this Opinion may be covered under this Opinion if the designation or expansion of that location is evaluated under a separate individual Section 7 consultation or through the Supersede review process outlined in Section 2.9.5 of this Opinion that will determine if placement in the new location has effects that are substantially similar to those analyzed in this Opinion.

New disposal areas must also meet all appropriate PDCs including (but not limited to):

- all placement, including ODMDS placement, do not create an obstruction of species movement in the area (e.g., does not create a mound that would deter or prevent species from moving through the area)(General PDCs in Section 2.2 in Appendix B),
- ensuring ODMDS locations meet the distance requirements in the PDCs from hardbottom used for turtles for foraging or shelter (General PDCs in Section 2.2 in Appendix B),
- ensuring ODMDS locations meet the distance requirements within the range of ESA-listed corals (Coral PDC Section 2 in Appendix C),
- and ensuring ODMDS locations meet the depth requirements within the range of Johnson's seagrass (Johnson's Seagrass PDC Section 2.1 in Appendix D).

2.4.5 Upland Placement (also referred to as Confined Disposal Facilities)

2.4.5.1 General Description of Upland Placement

Dredged material is sometimes transported to upland areas referred to as containment areas, upland disposal areas, confined disposal facilities, or dredged material management areas. These terms can be used interchangeably. Upland placement is contained within diked nearshore or upland confined disposal facility via hydraulic or mechanical means. A confined disposal facility is an engineered structure for containment of dredged material. The facility may be constructed as an upland site, nearshore site with 1 or more sides in water (sometimes called intertidal sites), or island containment area. Confined disposal facilities vary considerably in size, dike type, and method of filling. Although the volumes vary from year to year, the USACE estimates that 35% of the total volume of material dredged to maintain federal projects in the United States is placed in a confined disposal facility. The confinement or retention dikes or structures in a confined disposal facility enclose the placement area above any adjacent water surface, isolating the dredged material from adjacent waters during placement. These facilities are designed to retain as much of the fine-grained sediments as is practicable. This is typically managed through the use of weirs. The return flow of water (effluent) from a confined disposal facility is specifically defined as a discharge to waters of the United States under Section 404 of the Clean Water Act.

2.4.5.2 Upland Disposal Covered Under this Opinion

For the purposes of this Opinion, upland disposal is defined as occurring in an area outside of NMFS purview either on the uplands or a contained water body outside NMFS purview. The PDCs of this Opinion require that any return discharge water be maintained to prevent scouring or erosion of areas under NMFS purview. Upland placement with return waters can still generate turbidity even when confined by berms.

2.5 Types of Equipment and How Dredge Material is Transported

The equipment that is used to conduct activities covered under this Opinion may be used for dredging described in Section 2.3 of this Opinion, placement of material described in Section 2.4 of this Opinion, transportation of that material from the dredge location to the placement location, G&G surveys described in Section 2.6.3 of this Opinion, or during relocation trawling described in Section 2.7.1 of this Opinion. Each of these activity-based sections describe not only the activity (e.g., dredging, placement, or surveys), but also some of the activity-specific equipment that may be used (e.g., geophysical survey equipment or water-injection dredging methods) and a brief overview of the other types of equipment that may be used that is discussed in this section. This section provides more detail on the equipment types that may be used for multiple types of activities and why multiple types of vessels may operate simultaneously to achieve maximum efficiency and productivity.

The choice of equipment used is a balance between many factors including the cost to operate the machinery, effectiveness of the equipment in the specific project area, distance between the dredge and placement site, dredge material composition, and the risk of ESA-listed species encounters and effects. For example, all dredging activities described in Section 2.3 of this Opinion may be accomplished using mechanical or hydraulic dredging equipment described in this Section. However, generally, mechanical dredging is more commonly used for smaller projects and those in more confined areas while hopper dredging is more commonly used for larger navigation projects closer to beach nourishment projects or in open-water environments with the potential for rougher seas, as described in the descriptions in the mechanical and hydraulic dredging equipment sections below. Also, the use of hopper dredging, especially in certain locations may increase the risk of take of ESA-listed species whereas the use of certain mechanical equipment may increase the risk of turbidity and sedimentation, as briefly described by equipment type below and evaluated in the effects analysis in Section 3 of this Opinion.

Once the dredge location, size of the dredging project/ quantity of material intended to be removed, and intended use of the material dredged is determined, the equipment used to immediately relocate or transport the dredged material to another location is selected. A brief overview of how these equipment types may be combined to complete a project is provided in the list below. This list is intended to illustrate how equipment types may be used, and combined, acknowledging that other combinations of equipment may be used on a given project.

All activities and support equipment operation must adhere to the PDCs (Appendix B-Appendix H) of this Opinion and include equipment specific PDCs that were designed to reduce the risk of take of ESA-listed species, to reduce effects from turbidity and sedimentation generated during dredging or material placement based on the type of equipment used, and to preserve the ability for a protected species observer (PSO) to observe take based on different equipment types and modifications. New dredging technologies may be considered under the SARBO Supersede review process outlined in Section 2.9.5 of this Opinion.

- Mechanical dredging: Material dredged using mechanical dredging is scooped or lifted from the sea floor and may be deposited to either a location adjacent to the dredging (e.g., from a berth to the adjacent uplands) or loaded onto a barge and transported to another placement location.

- Hydraulic dredging: Material suctioned up from the sea floor during hydraulic dredging may be relocated by:

    - Pipeline: Pumping the material from a cutterhead or hopper dredge through a pipeline and depositing it to the intended location, such as an upland disposal site, beach placement site, or marsh creation area. Pipelines are used to transport materials either by pumping from a hopper dredge, a cutterhead dredge to a disposal location such as a beach, spoil area, or upland disposal site Figure 1, or hydraulic offloading out of a barge. These pipes can be placed on the sea floor or floated. Pipelines placed on the sea floor must either be of sufficient weight to remain in place or be anchored or weighted. Floating pipelines are anchored to the sea floor and may require booster pumps if the length of the pipeline is too long for the dredge to push the material to the placement location Figure 2. Pipelines are typically placed in the same pipeline corridor for each recurring event to minimize the potential damage to resources in the area. Additional PDC restrictions apply to the placement of pipelines within the range of Johnson's seagrass (Appendix D) and within the range of ESA-listed corals (Appendix C).

    - Split-hull: Emptying of a hopper dredge with a split-hull design where the material drops from the bottom of the vessel, through the water column, and settles in an area such as an ODMDS. An example of a split-hull hopper is shown in Figure 3.

        - Side-cast: Side-casting is used to disperse dredge material adjacent to the dredging site as shown in Figure 4.

- Barges/ scows: Barges or scows may be used to offload material from a barge using mechanical equipment such as a clamshell or bucket dredge to lift the material from the barge and deposit it into the water column to settle in an area such as an ODMDS. Scows are also used to transport dredged material to beach placement location and conveyed to the beach using a hydraulic offloader.

- Agitation Dredging: As described in Section 2.5.3 of this Opinion, agitation dredging such as bed-leveling or water-injection dredging may be used to directly move material out of the dredged location into the surrounding area or into the water column to be naturally transported down current.

- Geophysical surveys: During dredging or placement operations, a survey vessel may also be used to determine the resources present at the site (e.g., presence of hardbottom) or to determine of the appropriate dredge or placement depths are obtained.

- Relocation trawling: A trawling vessel may operate either prior to dredging to determine the potential presence of ESA-listed species in the area or prior to and/or concurrently with hopper dredging to intentionally capture ESA-listed species to relocate them out of the dredge area as a minimization measure to reduce take.
- Crew boats: Additional crew boats may also work current with vessels to transport needed crew or supplies to or from other larger vessels such as a hopper or relocation trawling vessel working on a 24-hour operation cycle for long periods of time.

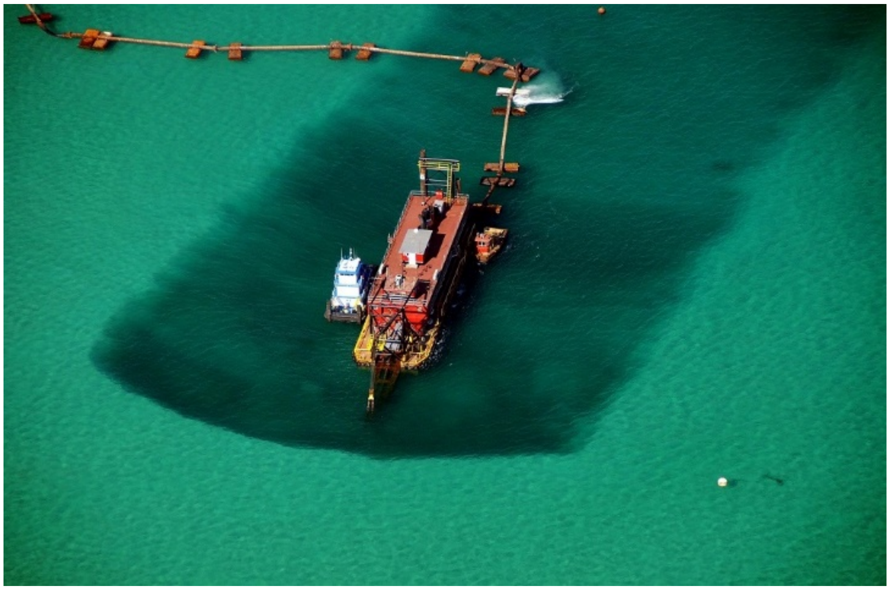

Figure 1. Floating pipeline from a cutterhead dredge
(Image provided on USACE Operations and Dredging Endangered Species System [ODESS] website https://dqm.usace.army.mil/odess/#/home).

Figure 2. Pipeline from a hopper dredge used for beach nourishment.
(Image provided by Great Lakes Dredging Company from the Egmont Key, Florida 2015 project).

Figure 3. Split Hull Hopper Dredge MURDEN.
(Image provided by USACE in SARBA)

Figure 4. Side-casting dredge MERRITT.
(Image provided by USACE in SARBA)

2.5.1 Mechanical Dredging Equipment

Mechanical dredging is a common dredging type that involves smaller, less expensive equipment that uses some form of bucket to excavate and raise the bottom material. Mechanical dredges remove material by scooping it from the bottom and then placing it onto a waiting barge or scow, or directly into a placement/disposal area. Mechanical dredges work best in consolidated, or hard-packed, materials and can be used to clear rocks and debris. Dredging buckets have difficulty retaining loose, fine materials, which can be washed from the bucket as it is raised. Special buckets have been designed for controlling the flow of water and material from buckets and are used when dredging contaminated sediments.

Mechanical dredges are rugged and can work in tightly confined areas. They are mounted on a large barge and are towed to the dredging site and secured in place by anchors or spuds. USACE reports that they are often used in harbors, around docks and piers, and in relatively protected channels, but are not suited for areas of high traffic or rough seas. These dredges can generate relatively large amounts of turbidity as the bucket traverses the water column.

2.5.1.1 Clamshell

Clamshell (aka bucket) dredges, named for the scooping buckets they employ, are the most common types of mechanical dredge (Figure 5). A clamshell dredge begins the digging operation by dropping the bucket in an open position from a point above the sediment. The bucket falls through the water and penetrates into the bottom material. The sides of the bucket are then closed and material is sheared from the bottom and contained in the bucket compartment. The bucket is raised above the water surface, swung to a point over the barge, and then released into the barge by opening the sides of the bucket. Usually 2 or more disposal barges, called dump scows, are used in conjunction with the mechanical dredge. While 1 barge is being filled, another is being towed to the dumpsite by a tug and emptied. If a diked disposal area is used, the material must be unloaded using mechanical or hydraulic equipment. Using numerous barges, work can proceed continuously, only interrupted by changing dump scows or moving the dredge. This makes mechanical clamshell dredges particularly well suited for dredging projects where the disposal site is many miles away.

Figure 5. Mechanical clamshell dredge.
Photos provided by the USACE in SARBA.

2.5.1.2 Backhoe

Backhoe dredges operate by scooping material from the bottom and placing in a waiting barge or into a disposal area. The backhoe dredge uses a bucket that is structurally connected to the dredge by the rigid member configuration as shown in Figure 6. To increase digging power, the dredge barge is moored on powered spuds that transfer the weight of the forward section of the dredge to the bottom to provide reaction forces to the digging-induced forces. The maximum bucket size that can be used for a specific project depends on the rated capacity of the excavator, sediment characteristics, and water depth. Bucket sizes generally range from 6 to 25 yards (0.6-19 m). Larger backhoes can excavate to a maximum depth of approximately 80 ft (24 m). The density of sediment excavated can almost equal its in situ density but, like other conventional mechanical dredges, it may generate a relatively large amount of sediment resuspension at the dredge site.

Figure 6. Backhoe Dredge NEW YORK.
Photo courtesy of Great Lakes Dredge and Dock Company, Oak Brook, IL, provided by USACE in SARBA.

2.5.2 Hydraulic Dredging Equipment

Hydraulic dredging is characterized by the use of a centrifugal pump to dredge sediment and the transportation of the dredged material slurry and water to identified discharge areas. The ratio of water to sediment within the slurry mixture is controlled to maximize efficiency. Too little water and the dredge will bog down; too much and the dredge won't be efficient in its work and it will take longer to dredge the shoals. These suction type dredging methods result in decreased turbidity and sedimentation concerns, though turbidity can still be a concern from overflow of hopper dredges and scows or improperly sealed pipelines. The types of hydraulic dredges used by USACE and/or BOEM are cutterhead pipeline and hopper dredges discussed below.

2.5.2.1 Cutterhead Suction/ Pipeline Dredging

Cutterhead pipeline dredges are designed to handle a wide range of materials including clay, hardpan, silts, sands, gravel, and some types of rock formations without blasting. They are used for new work and maintenance in projects where suitable placement/disposal areas are available and operate in an almost continuous dredging cycle resulting in maximum production, economy, and efficiency. A cutterhead is a mechanical device that has rotating blades or teeth to break up or loosen the bottom material so that it can be sucked through the dredge pipeline (Figure 7).

Cutterhead pipeline dredges are rarely self-propelled, and typically must be transported to and from the dredge site where they are secured in place by special anchor pilings, called spuds. Pipeline dredge size is based on the inside diameter of the discharge pipe which commonly ranges from 6- to 36-inches. Cutterhead pipeline dredges are capable of dredging in shallow or deep water and have accurate bottom and side slope cutting capability. They require an extensive array of support equipment including pipeline (floating, shore, and submerged), boats (crew, work, survey), barges, and pipe handling equipment. Most cutterhead pipeline dredges have a cutterhead on the suction end. Limitations of these dredges include relative lack of mobility, long mobilization and demobilization, inability to work in high wave action and currents, and they are impractical in high traffic areas.

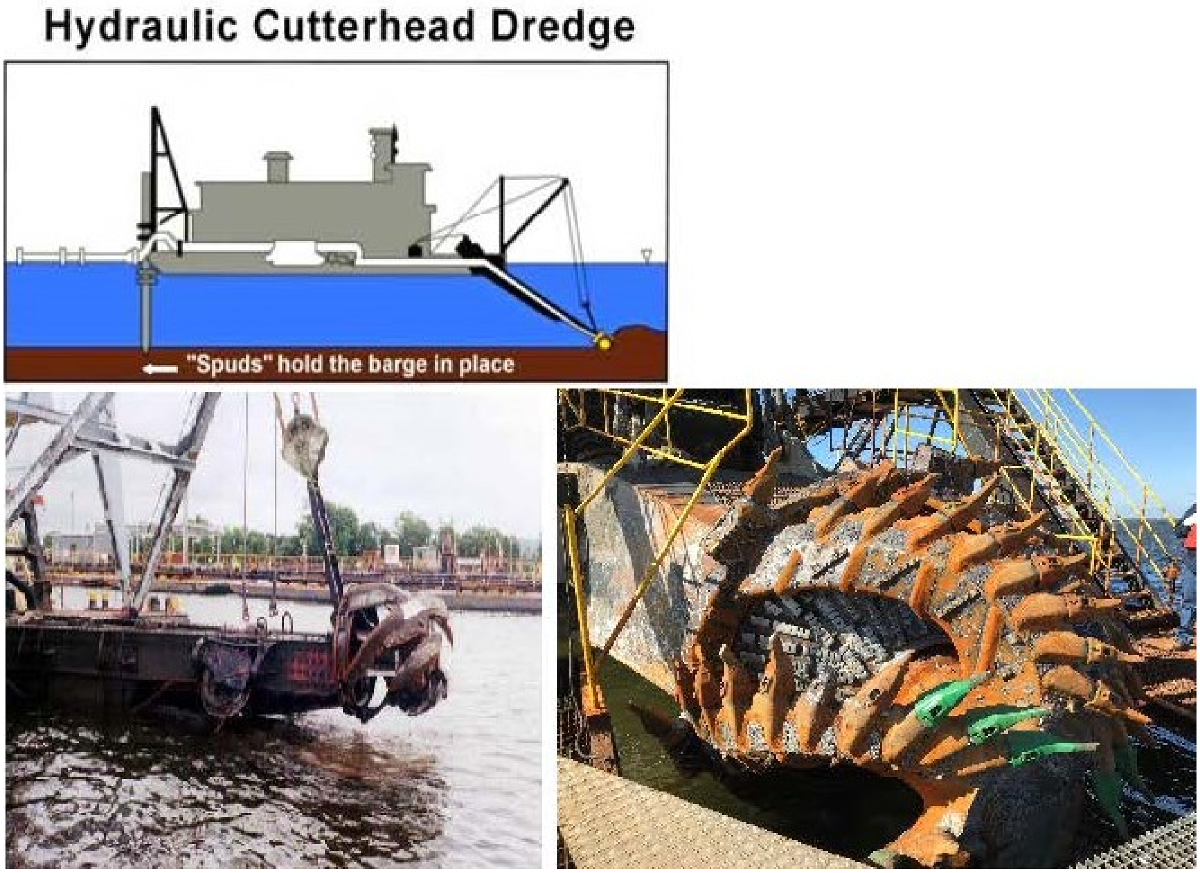

Figure 7. Cutterhead pipeline dredge schematic shown on the top and 2 representative close-up photographs below to show the variety in size of cutterhead dredges[10].

During the dredging operation a cutterhead suction dredge is held in position by 2 spuds at the stern of the dredge, only one of which can be on the bottom while the dredge swings. Some cutterhead pipeline dredges use a system of anchors and winches to hold themselves in place and/or advance forward. There are 2 swing anchors some distance from either side of the dredge, which are connected by wire rope to the swing winches. The dredge swings to port and starboard alternately, passing the cutter through the bottom material until the proper depth is achieved. The dredge advances by "walking" itself forward on the spuds. This is accomplished by swinging the dredge to the port, using the port spud and appropriate distance, then the starboard spud is dropped and the port spud raised. The dredge is then swung an equal distance to the starboard and the port spud is dropped and the starboard spud raised.

Cutterhead pipeline dredges work best in large areas with deep shoals, where the cutterhead is buried in the bottom. A cutterhead removes dredged material through an intake pipe and then pushes it out the discharge pipeline directly to the placement/disposal site. Most, but not all, pipeline dredging operations involve upland placement/disposal of the dredged material. Therefore, the discharge end of the pipeline is connected to shore pipe. When effective pumping

[10] The top and bottom left photos provided by the USACE in SARBA and the bottom right photo provided by Nicole Bonine of NMFS from a dredge tour of the Carolina cutterhead in Tampa Bay on November 17, 2018.

distances to the placement/disposal site become too long, a booster pump is added to the pipeline to increase the efficiency of the dredging operation. Though not common, cutterhead pipeline dredges may be used on offshore dredging projects where the placement distance exceeds the capabilities of booster pumps. Specifically, the cutterhead pipeline dredge is used in combination with a spider barge/scow operation and transported by tugs to a hydraulic off-loader located just offshore of the placement site (e.g. Caminada Headland Project, Gulf of Mexico).

In most cases material is pumped directly from the dredged area to a placement/disposal site including using a pipeline to transport the dredged material to an upland location or a barge for transport to a hydraulic off-load site. As such, there is no opportunity to monitor for biological material on board the dredge. Monitoring at the placement/disposal site is also challenging due to the volume of material pumped, often to the uplands, and often unsafe for an observer. Considering that the cutterhead is typically buried in the sediment to promote operational efficiency; thus, limiting exposure in the water column to the suction field, cutterhead dredging has historically resulted in significantly lower takes of ESA-listed species than hopper dredging.

2.5.2.2 Hopper

The hopper dredge, or trailing suction dredge, is a self-propelled ocean-going vessel with a section of the hull compartmented into 1 or more hoppers. Fitted with powerful pumps, the dredges suck sediment from the surface of the seafloor through long intake pipes, called dragarms, and store it in the hoppers. Normal hopper dredge configuration has 2 dragarms, one on each side of the vessel. A dragarm is a pipe suspended over the side of the vessel with a suction opening called a draghead for contact with the bottom (Figure 8). Depending on the hopper dredge, a slurry of water and sediment is generated from the plowing of the draghead "teeth," the use of high pressure water jets, and the suction velocity of the pumps. The dredged slurry is distributed within the vessels hopper allowing for solids to settle out and the water portion of the slurry to be discharged from the vessel during operations through its overflow system. When the hopper attains a full load, dredging stops and the ship travels to either an in-water placement site, where the dredged material is discharged through the bottom of the ship by splitting the hull, or opening doors in the bottom of the hull, or hooks up to an in-water pipeline, where the dredged material is transported to a shore placement site (e.g., beach nourishment).

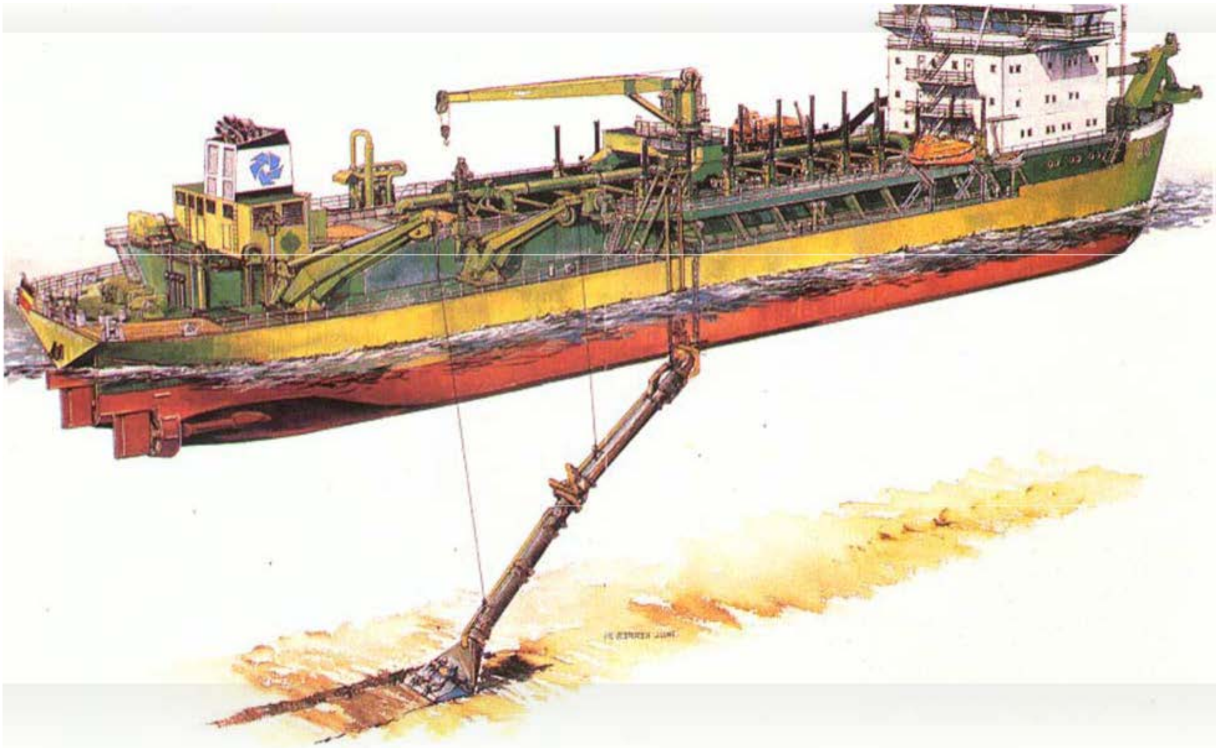

Figure 8. Hopper dredge illustration. Image provided by USACE in SARBA.

Hopper dredges are well suited to dredging heavy sands. They can work in relatively rough seas but safety, effectiveness, and costs are a concern. Because they are mobile, they can be used in high-traffic areas. They are often used at ocean entrances and offshore, but cannot be used in confined or shallow areas due to their size and draft.

Hopper dredges can move quickly to disposal sites under their own power (maximum speed unloaded ≤ 17 knots; maximum loaded ≤ 16 knots), but since the dredging stops during the transit to and from the disposal area, the operation loses efficiency if the haul distance is too far. Based on the review of hopper dredge speed data provided by the USACE Dredging Quality Management program, the average speed for hopper dredges while dredging is between 1-3 knots, with most dredges never exceeding 4 knots (Jay Rosatti, USACE Engineer Research and Development Center [ERDC], personal communication). PDCs in this Opinion require slower transit speeds to disposal sites when North Atlantic right whale are present in the action area (Appendix F).

Hopper dredges also have several limitations. Considering their normal operating conditions, hopper dredges cannot dredge continuously unlike other dredge types that continue to work and transfer dredged material to another location. Hopper dredges must stop dredging while transporting materials to the final destination. The precision of hopper dredging is lower than other types of dredges; therefore, they have difficulty dredging steep side banks and cannot effectively dredge around structures. For example, dragheads may “crab” or move under or onto side slopes as a result of bottom conditions, bottom currents, or location of the dredge in or near the side of the channel. Crabbing may result in dragheads not being maintained on the bottom due to the more frequent need to pick up and realign the dragarms. Therefore, there is an increased risk of sea turtle entrainment when dredging within environments that may result in a higher risk of crabbing.

Hopper dredges also vary in total size and draghead size. Smaller “modified” hopper dredges such as the CURRITUCK and MURDEN, have historically not resulted in entrainment of ESA-listed species and hence have had fewer restrictions than larger, traditional hopper dredges. Their small size and operating characteristics including small draghead sizes (2-ft by 2-ft, to 2-ft by 3-ft), small draghead openings (5-inch by 5-inch to 5-inch by 8-inch), small suction intake pipe diameters (10-14 inch), and limited draghead suction (350- 400 horsepower) result in a lower suction force that sea turtles are believed to be able to outswim. In 1999 NMFS reviewed these CURRITUCK type of hopper dredges and determined that a sea turtle deflector shield, draghead screening, and protected species observers were not needed due to the low probability of entrainment and no reports of take (NMFS 1999). The USACE confirmed in October 2018, that they still do not have any records of take associated with these smaller draghead and low suction velocity types of hopper dredges.

2.5.2.2.1 Draghead Deflectors

In order to minimize the risk of incidental takes of sea turtles, sea turtle deflectors are added to the dragheads used on hopper-dredging projects where the potential for sea turtle interactions exist (discussed as a PDC) and the dredging environment does not reduce the efficacy of the deflector or increase the risk for sea turtle interaction (Figure 9). The leading edge of the deflector is designed to have a plowing effect of at least 6-inch depth when the drag head is being operated. Appropriate instrumentation is required on board the vessel to ensure that the critical “approach angle” is attained in order to satisfy the 6-inch plowing depth requirement (USACE 1993).

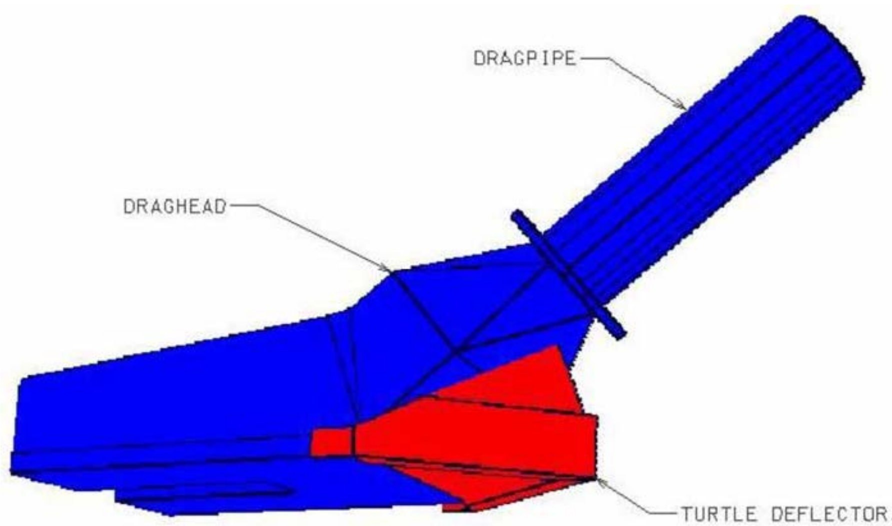

Figure 9. Illustration of a hopper dredge draghead with installed sea turtle deflector.

2.5.2.2.2 Hopper dredging screening

Screening is used during hopper operations to either exclude certain materials from entering the draghead or retain entrained ESA-listed species within the hopper inflow box or overflow screening for observation and reporting by PSOs. Screening of the draghead, inflow box, and overflow is used for different purposes and may require different specifications as described below:

Draghead screening
In areas with the potential for entraining Munitions of Explosive Concern (MEC) or larger sized incompatible material (i.e., shell, rock, etc.), a smaller mesh screening may be installed to the bottom of the draghead to exclude these items from entering the hopper (Figure 10). Screening used for this purpose is often referred to as MEC or unexploded ordinance (UXO) screening and typically consists of longitudinal bars with opening/ spacing of 1.25 - 1.5 in by 6 in on the dragheads. The dimensions of the screen bars are designed and constructed in a manner to exclude undesirable material while maximizing the total open area of the suction head through which sand can be dredged and maximize the hydraulic transport efficiency of the draghead. This smaller screening size may exclude ESA-listed species from entering the inflow box on board the hopper dredge; thus limiting the ability for PSOs to identify and report species that may have been taken by the operation. Though draghead screening may exclude the ability of PSOs to detect ESA-listed species within inflow and overflow screening, it does not limit the risk of ESA-listed species impingement and mortality.

Draghead screens may clog and require cleaning, which can be completed by raising the dragarm so that the flow of water removes items or by raising the dragarm to the deck of the vessel to be manually cleaned. The PDCs of this Opinion prohibit cleaning of the dragarm by rinsing the draghead in the water while running the pumps as this may increase the risk of entrainment.

Figure 10. Draghead with (left) and without (right) UXO Screening[11]

[11] Image on left shows a draghead with UXO screening [photo from a USACE presentation on Beach Replenishment Operational Challenges by Paul Green, USACE Baltimore District]. The image on the right shows a draghead without USXO screening [provided by Karla Reece of NMFS from a dredge tour of the Terrapin Island hopper dredge in Tampa Bay on November 17, 2018]

Inflow screening
Once material enters the drag head by suction generated at the pump positioned along the draghead arm, a slurry of water and sediment material passes through a screened inflow box on its way to the hopper (Figure 11). Generally, screening has 4-inch by 4-inch openings to optimize the inflow of material while still ensuring accountability of entrained species. The purpose of the inflow screening is for PSOs to monitor for entrained protected species and bycatch; however, other debris (i.e., rock, clay, wood, trash, etc.) larger than the screen size may also be captured resulting in the potential for clogging the boxes. For example, dredging projects in Wilmington Harbor have frequently encountered large debris requiring the temporary increase in screening size or removal of screening altogether until the debris of concern has been removed from the channel. Changing the size of screening requires welding on a new screen and takes time so limiting the number of changes in screen sizes is important.

Figure 11. Images of various inflow boxes that shows the variety in size and screening[12]

Overflow screening
The dredged material slurry that collects in the hopper is dewatered by allowing for the water to overflow out of the hopper while coarser sediment is retained (Figure 12). Before the overflow water is released, it passes through an overflow screen to ensure additional observation and reporting of entrained species. This overflow may or may not be screened depending on the hopper dredge.

[12] The left 2 images were provided by Mark Dodd of the Department of Natural Resources Wildlife Resource Division and the third image was provided by Nicole Bonine of NMFS from a dredge tour of the Terrapin Island_ hopper dredging in Tampa Bay on November 17, 2018.

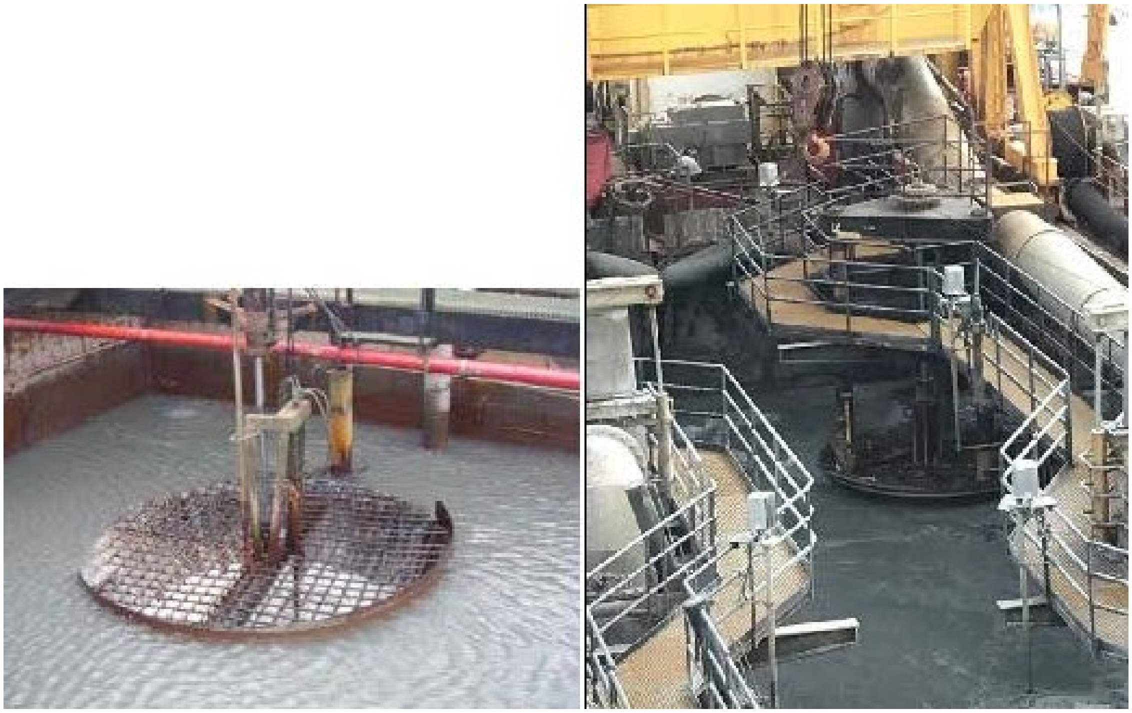

Figure 12. 2 examples of overflow screening[13]

2.5.3 Agitation Dredging

2.5.3.1 General Description of Agitation Dredging

Agitation dredging is a process that intentionally discharges dredged material into the water column instead of using another piece of equipment to move it out of the dredging location under the assumption that a major portion of the sediments will be transported and permanently deposited outside the channel prism by tidal, river, or littoral currents. Agitation dredging is typically used only when there are currents in the surrounding water to carry the sediments from the channel, and when the risk to environmental resources is low. Favorable conditions may exist at a particular project only at certain times of the day, such as at ebb tides, or only at such periods when the stream-flow is high. To use agitation dredging effectively requires extensive studies of the project conditions and definitive environmental assessments of the effects. Agitation dredging is not typically performed in slack water or when prevailing currents permit redeposit of substantial quantities of the dredged material in the project area or in any other area where future excavation may be required. For the purpose of this Opinion, agitation dredging includes bed-leveling or water injection dredging described below and limited by the PDCs in Appendix B.

[13] The left image was provided by Mark Dodd of the Department of Natural Resources Wildlife Resource Division and the right image provided by Nicole Bonine of NMFS from a dredge tour of the Terrapin Island hopper dredging in Tampa Bay on November 17, 2018

2.5.3.1.1 Bed-leveling

A “bed-leveler” is considered to be any type of dragged device used to smooth sediment bottom irregularities left by a dredge (Figure 13). It is also referred to as a “mechanical leveling device or drag bar”. In various parts of the United States this process is known as “barring” or “knockdown” (Engineer Research and Development Center 2003). Use of bed-levelers can be documented as far back as 1565 (van der Graaf 1987). Typically, a bed-leveler consists of a large customized plow, I-beam, or old spud that is slowly dragged across the sediment. It can be used either to smooth out peaks and trenches during the final cleanup phase of the dredging activity or as the primary form of dredging used to redistribute sediments to maintain navigable depths rather than removing them by dredging with conventional methods.

Bed-leveling used during the final/clean up phase of dredging, is done by dragging the drag bar to knock down and even out the bottom sediment caused by other forms of dredging. Bed-leveling is sometimes also used as the primary form of dredging to drag a thin layers of material out of the project area or to knock down high points in a project area in between dredging cycles. For example, material that has accumulated in a berth along a river may use bed-leveling to move the material back into the main channel of the river. Another example is to level out the high points within an in-water disposal area so that more material can be placed while remaining within the approved height of the disposal area.

The design of a bed-leveler, and how it connects to the chains used to drag it, can create pinch points where an animal can be impinged, as shown in Figure 14. Also shown in Figure 14 is how modifications to the bed-leveler can reduce that risk. All bed-leveling covered under this Opinion must follow the PDCs, including the bed-leveling specific PDCs in Section 3.4 of Appendix B that addresses concerns about pinch points. Bed levelers used under this Opinion must also be of a design that creates a “sand wave,” which is understood to cause ESA-listed species to move away from the equipment.

Figure 13. Example bed-levelers (USACE 2015a)

Figure 14. The image on the left shows a bed-leveler design with the attachment points extending beyond the side of the blade leading to a potential pinch point. The image on the right shows additional bade being welded in place to eliminate the pinch point (USACE 2015a).

2.5.3.1.2 Water Injection Dredging

Water injection dredging (WID) is a method that is similar to a bed-leveling system in that it drags a bar behind a barge. For this method, high volumes of low pressure water (approximately 2-2.5 ft per second) are pumped through a series of nozzles on a wide horizontal jetbar directly into the bottom sediments. This disrupts the internal friction and cohesive properties of the sediment to create a fluid mud layer that remains close to the bottom and is washed away by the outgoing tide until settling out further downstream. Unlike other dredging methods, gravity and water currents maintain the fluid mud created by WID within the bottom portion (usually within the bottom 2 meters [m]) of the river and does not create turbidity in the middle and upper layers. The turbidity plume narrows as it moves downstream due to the gravitational force of the river as shown in Figure 15 below. Water is pumped from higher in the water column and the intake is screened to minimize entrainment. Like bed-leveling, WID is designed to move thin layers of sediment out of the project footprint.

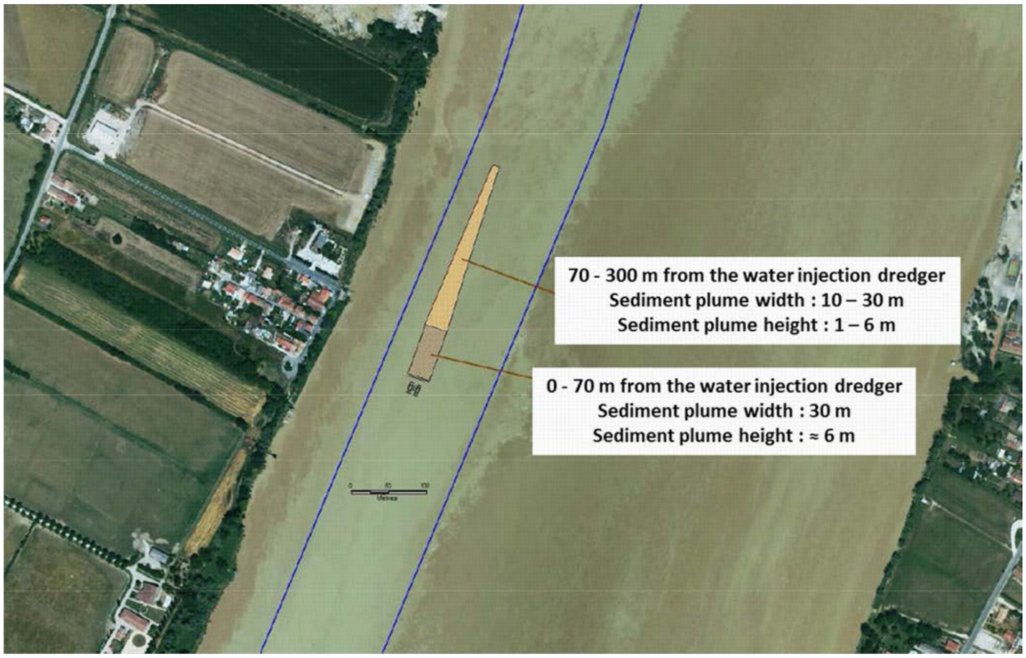

Figure 15. Spatial extent of the density current in the Gironde estuary (Ginger 2012)

2.5.3.2 Agitation Dredging Covered Under this Opinion

For the purposes of this Opinion, agitation dredging is limited to the list of activities provided below that follow all relevant PDCs in this Opinion.

- Bed-leveling and water-injection dredging used as a form of maintenance dredging in areas covered under this Opinion and within the previously authorized or permitted dredge footprint to the analyzed dredged template, as described in Section 2.5.3 of this Opinion.
- Bed-leveling and water-injection dredging used as a form of maintenance dredging in areas that have undergone an individual Section 7 consultation and require a repeat dredging event to return the area to the analyzed dredge template.
- Additional equipment specific bed-leveling PDCs are provided in the General PDCs in Section 3.4 of Appendix B.

Additional PDCs limit or prohibit bed-leveling and water-injection dredging within the range of Johnson's seagrass (Johnson's Seagrass PCDs, Appendix D) and within the range of ESA-listed corals (Coral PDCs, Appendix C) that were designed to protect seagrasses and corals/coral hardbottom from burial from the resuspended sediments generated using this dredging method.

2.5.4 Support Vessels

Depending on the dredging and placement site conditions for an individual project, accompanying equipment such as tugs, barges, crew transport vessels, and/or survey vessels may be used in association with dredging activity. Methods of transporting dredged material to placement sites include self-propelled transport via hopper dredges, towing of loaded barges to placement sites via tugboats, or pipelines connecting dredging and placement locations. Tugboats are a component of most dredging operations and may be used to move immobile equipment into place as well as towing loaded barges to the placement sites.

Survey vessels may be used to complete G&G surveys discussed below. These often include large vessels that can be used to support multiple dredging projects simultaneously resulting in frequent high speed trips between the dredge location and placement area for each dredging project and multiple trips between different projects (e.g., Mayport, Florida and Tybee Island, Georgia). Currently, the USACE's primary geophysical offshore survey vessel off the coast of Florida and Georgia is the Florida II, a 62 ft catamaran style vessel with speeds up to 36 kts (Figure 16). The North Atlantic Right Whale Conservation Plan (Appendix F) limits the maximum speed of support vessels when and where North Atlantic right whales may be present, as defined in Appendix F.

Figure 16. Image of the USACE Survey Vessel (Florida II).
Image provided on MarineTraffic.com.

2.6 Geophysical and Geotechnical Surveys

2.6.1 General description of G&G surveys

Geotechnical surveys are used to collect sediment samples, while geophysical surveys utilize scanning equipment to determine the substrate including the presence of bottom features such as hardbottom or cultural resources and to determine pre- and post- construction bathymetry. G&G surveys are performed to evaluate those geologic, geotechnical, and soil conditions that affect the safety, cost effectiveness, design, and execution of a proposed coastal storm risk management project or navigation maintenance dredging project. There are different methods used for coastal storm risk management projects and maintenance dredging projects. The purpose of conducting a G&G survey for a coastal storm risk management project is to locate offshore sand borrow sources, which are of sufficient size, quality, and proximity to the shoreline to make construction economically feasible and environmentally acceptable. Additional purposes include ensuring that borrow areas do not contain, or may adequately avoid, submerged shipwrecks, other historic/prehistoric resources, and hardbottom habitat. Navigation dredging projects require G&G surveys that are tailored to evaluate surface or subsurface conditions for both maintenance dredging and beneficial use of dredge material or new borrow source identification, evaluation of new work material for a harbor deepening/expansion, or to evaluate work that have been completed.

2.6.2 G&G Surveys covered Under this Opinion

G&G surveys may be undertaken or authorized by the USACE for Civil Works or Regulatory projects when deemed necessary to complete dredging and material placement projects covered under this Opinion and as limited by the PDCs. BOEM has completed a separate consultation with NMFS that reviewed their G&G survey program associated with the Marine Minerals Program, as explained in Section2.5.4 of Appendix K as part of the additional consultation history.

Geophysical surveys covered under this Opinion are limited to only electromechanical sources including boomers, chirp sub-bottom profilers, side-scan sonars, and single beam, interferometric, or multibeam depth sounders operated at the lowest power setting, narrowest beamwidth, and highest frequency possible to fulfill data needs and to effectively reduce exposure and received sound levels. Examples of equipment used by the USACE for geophysical surveying are provided in Table 4, but must be operated consistent with the G&G PDCs in Appendix G to be covered under this Opinion. No airguns or other deep-penetrating geophysical instruments such as sparkers are allowed under 2020 SARBO. In addition, survey vessels used to complete this work must adhere to the speed restrictions outlined in the North Atlantic Right Whale Plan (Appendix F) when working in the area and during the time of year when North Atlantic right whales may be present.

2.6.3 Geophysical Surveys

Geophysical surveys covered in this Opinion typically use a high-resolution sound source and receiver system towed behind a vessel. The types of equipment typically used for these surveys consist of sub-bottom profilers, side-scan sonars, magnetometers, and single and multibeam bathymetry sonars. The acoustic signals produced by these systems are usually impulsive, tonal, or chirp pulses (short-duration signals that sweep through many frequencies).

Table 4 provides examples of geophysical survey equipment used by the USACE for projects covered under this Opinion, which must be operated according to the PDCs in Appendix G to be covered under this Opinion. If the USACE intends to use any additional equipment that does not meet all of the PDCs in this Opinion, then the use of that equipment would require an evaluation under the SARBO Supersede procedures outlined in Section 2.9.5 of this Opinion. Information gathered on the potential acoustic effects of geophysical surveys is described in the additional consultation history information provided Section 2.5.4 of Appendix K.

Table 4. Examples of Geophysical Survey Equipment Provided by USACE

Type of Sound Source	Equipment Name	Source Level (dB re 1 Pa at 1 m)[14]	Pulse Durations – microsecond	Operating Frequencies - kilohertz (kHz)
Side-scan sonar	EdgeTech 4200	218 (210-226) dB (rms)	0.6-26	6, 105, 200, 210, 240, 410, 540, 1,600
Side-scan sonar	Klein 3900	249 dB (rms)	200	455-900
Side-scan sonar	Klein 3000	234 – 242 dB	25–400	135 - 445
Multibeam Echosounder	Simrad EM 2000	207 dB (rms) 218 dB (peak)	0.2	200
Multibeam Echosounder	Kongsberg EM 2040	208 dB (rms)	0.2	200-400
Multibeam Echosounder	Reson 7125	223 dB (rms)	33-300	200-400
Multibeam Echosounder	R2 Sonic 2024	221 dB (rms)	Unavailable	200-400
Multibeam Echosounder	Reson 7111	223 dB	0.8 -500	100
Multibeam Echosounder	Reson T20-P	200W / 300W	30-300 (CW) 0.3–10 (FM)	200 - 400
Single Beam Echosounder	Odom CV200	203dB (rms)	5	24-200
Single Beam Echosounder	Teledyne Odom Hydrotrac	152 dB (rms)	0.01 at 24 kHz, 0.1 at 200 kHz	24-340
Boomer	Applied Acoustics 251	212 dB	120 – 180	0.1 - 5
Boomer	Applied Acoustics 301	215 dB	200	0.1 – 7.5
Boomer	Applied Acoustics 252	212 dB	200	0.1 - 5
Sub-bottom profiler	Knudsen 3202	209 dB (estimated)	0.63–64	3.5
Sub-bottom profiler	EdgeTech DW-216	160 dB (rms)	20	2-16
Sub-bottom profiler	EdgeTech DW106	216 dB (rms)	40	2-6
Sub-bottom profiler	EdgeTech SB512i	212 dB (rms)	5-50	0.5-12
Sub-bottom profiler	Geopulse	186 (pulse to pulse)	20	3.5

[14] Source Level: Decibel (dB) relative to (re) 1 Pascal (Pa) at 1 meter (m)

2.6.3.1 Sub-bottom Profiling Sonar

Sub-bottom profiling employs the use of chirp and/or boomer systems, which delineate near-surface geologic strata and features. These systems use a transducer to produce a sharp repeatable impulse and receive the return of the pulse once it is reflected off of the seafloor. Boomers are impulsive signals, which are broadband with most energy at low frequencies, and chirp sonars ("chirpers") emit chirp pulses (mentioned above) – frequency sweeps with most energy at high frequencies. Chirp systems are typically towed at depth, a certain distance off the seafloor. Boomers are towed at or near the surface.

Figure 17. Sub-bottom profiler.
Image provided by USACE in SARBA(USACE 2017).

2.6.3.2 Hydrographic Surveys (multibeam and single beam echosounders)

Hydrographic surveys include both single and multibeam echosounders that collect bathymetry data. The systems also detect acoustic backscatter which is used to characterize the seabed to aid in archaeological, benthic and sediment composition surveys. These types of equipment emit high-frequency tones or chirp signals.

2.6.3.3 Side-Scan Sonar

A side-scan sonar survey provides a higher level of detail during the reconnaissance phase of an investigation. Side-scan sonar generates an image of seabed morphology, submerged objects, and other features. (Figure 18). Changes in backscatter intensity generally result from changes in sediment composition and texture, presence of hardbottom/ledges, archaeological resources / shipwrecks, debris, etc. It may be used to infer zones of coarse and fine grained materials; however, physical sampling is required in order to properly characterize the material.

Figure 18. Photo of a side-scan sonar.
Image from Coastal Carolina University provided by the USACE in SARBA.

2.6.3.4 Magnetometer Remote Sensing

The marine magnetometer is a passive remote sensing device (i.e., nothing is emitted) that identifies materials with ferrous or ferric components or other objects having a distinct magnetic signature (Figure 19). This method is commonly used in underwater archaeological surveys to identify any potential historic resources. It has also been used in navigational projects to identify submerged wrecks, debris, pipelines, and utilities, and in some instances, UXO.

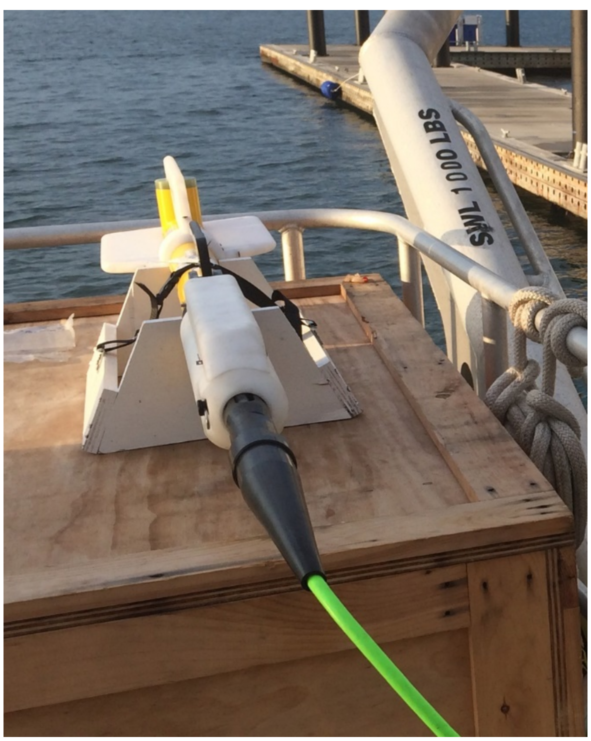

Figure 19. Photo of a magnetometer.
Image from USACE.

2.6.4 Geotechnical Surveys/Sediment Characterization

Geotechnical surveys (coring and sediment sampling) are most frequently performed concurrent with, or after geophysical surveying. The methods chosen are dependent upon the type of project and the engineering and environmental design and survey requirements. For coastal storm risk management projects, these surveys are typically conducted to identify and characterize the sediment volume, quality, and geological characteristics of a prospective borrow area. For established navigation projects requiring regular maintenance dredging, a geotechnical survey may be conducted to locate and evaluate the volume and quality of sediments in a shoal for a variety of reasons. In the case of a harbor deepening or expansion, an extensive geotechnical survey is conducted to delineate the subsurface soils and rock within the proposed dredging prism, determine the best method of removing the materials, and determine the storage capacity and/or develop engineering recommendations for existing disposal areas. The following techniques are the most commonly used methods for USACE geotechnical surveys and sediment characterization methods related to navigation dredging and coastal storm risk management projects:

- Geological borings
  - Vibracoring
  - Standard penetrometer testing
  - Wash/jet probing
- Surficial grab sampling

Nearly all geotechnical sampling occurs from either survey vessels or work barges towed into place. Some operational platforms require anchoring for brief periods with small anchors. Sometimes jack-up barges and spudded work barges are used. Surveys typically last only a few days and disturb a minimal area of seabed during individual sampling events (e.g., collection of a core or grab sample). Although vibracoring is the most likely technique used, other sampling methods such as piston or box coring and jet probes are also used as part of geotechnical surveys. Geological sampling disturbs the seafloor; however, due to the small size of the cores and platforms, the area of seabed to be disturbed during individual sampling events is minor. Under this Opinion, geotechnical surveys are used in areas where dredging is allowed under the PDCs and must be performed in accordance with the PDCs to be covered under this Opinion.

2.6.4.1 Geologic Borings

Geological borings are collected to describe the basic geologic materials including surface and subsurface sediments for engineering analyses on dredging material for navigation or beach nourishment projects. Many different types of borings can be performed including auger borings, drive borings, standard penetration test borings, washprobes, cone penetration tests, vibracoring, and rock core boring. The following sections provide information on the most commonly used techniques for navigation projects and borrow site surveys.

2.6.4.2 Vibracoring

A 3- or 4-inch diameter aluminum core barrel mounted on a platform or support assembly would be used to penetrate sediments in the upper 20 ft (6 m) of the seafloor. A sediment sample of 5-20 ft (1.5 to 6 m) would be acquired to determine sediment characteristics and sand resource thickness. To penetrate seafloor sediments, the core barrel is vibrated by a pneumatic or electric vibrahead, which results in local liquefaction of sediment along the core barrel surface, facilitating penetration into the sediment. Depending on local conditions, a typical vibracore survey can obtain 15-25 cores approximately 20 ft (6 m) deep in an area measuring 1 mi^2 per day.

2.6.4.3 Standard Penetrometer Tests

The Standard Penetration Test is described by the American Society for Testing and Materials ASTM[15] Standards as a test procedure by which a splitspoon sampler is driven, using a known energy, to obtain a representative soil sample for identification purposes, and to measure the resistance of the soil to penetration (compactness). The driving energy is imparted to the sampler (and length of drill rod) from the blows of a 140-lb (pound) hammer free-falling 30-inches. The test provides an indication of the relative density of granular soils, such as sand and gravel (Figure 20). The test method is used extensively to quantify soil properties for geotechnical engineering design.

[15] https://www.astm.org/Standards/D1586.htm

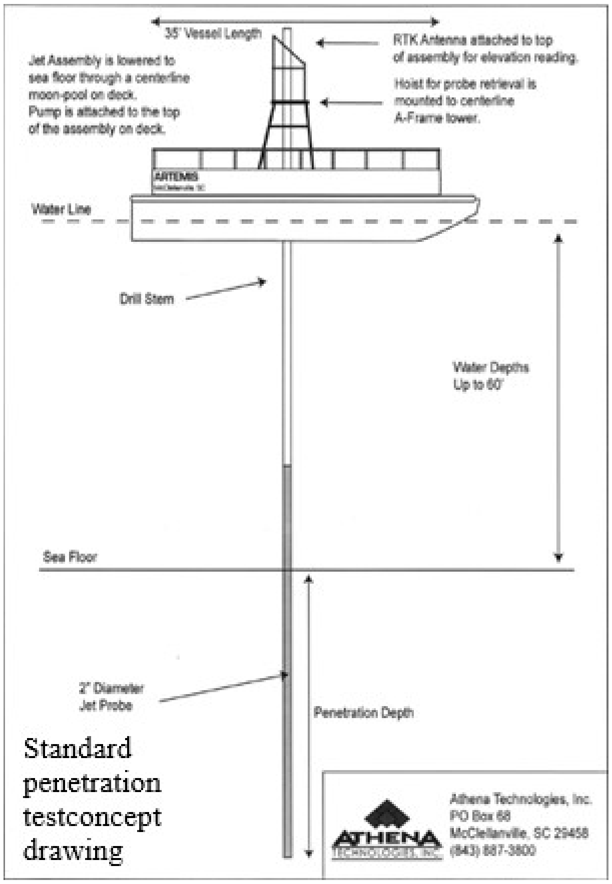

Figure 20. Drawing of a Standard Penetration Test.
Image provided by USACE in SARBA (USACE 2017).

2.6.4.4 Wash/Jet Probing

For this technique, wash probes or jet probes are advanced into the seafloor using a 1.5-inch hollow steel probe, 2-inch steel drill stems, and a 3-inch flexible hose connected to a water pump aboard the work vessel. The probe, pipe, and hose are connected via reducers and cam-lock pipe fittings. The operator lowers the wash/jet probe to the seafloor. Once the probe strikes the seafloor, the water pump is turned on, resulting in a blast of water emerging from the tip of the probe. The probe is advanced under its own weight until refusal is encountered. Upon refusal, the penetration depth is recorded and the probe is retrieved using a mechanical winch system. Wash probing is most commonly used in navigation projects to rapidly ascertain the presence of rock or stiff/dense material within a proposed dredge prism.

2.6.4.5 Grab Sampling

Grab samplers collect samples of the topmost layers of the seabed and benthic biota by bringing 2 steel clamshells together and cutting a bite from the soil. The grab sampler consists of 2 steel clamshells on a single or double pivot brought together either by a powerful spring or powered hydraulic rams operated from the support vessel (Figure 21). The grab is lowered to the seabed and activated either automatically or by remote control and the sample is raised to the vessel for examination. Typically, 3-4 grabs can be obtained per hour, but is dependent on specific equipment, sample depth, sediment type, and distance between samples.

Figure 21. Photo of a Grab Sampler.
Image provided by USACE from Eco Environmental of a Ponar grab sampler.

2.7 Monitoring for and Handling of ESA-listed Species Encountered During Projects Covered Under this Opinion

Monitoring for and handling of ESA-listed species encountered during activities covered under this Opinion is allowed, and in some instances required, as part of the minimization measure to minimize the impacts of nonlethal take and the amount of lethal take of ESA-listed species. Appendix H describes the procedures for monitoring for the presence of ESA-listed species in the area to avoid collision, handle species captured alive or dead, and how to handle those captured during the project. Additional information is provided below on relocation trawling, ESA-listed species handling and data collection, and monitoring by aerial surveys.

2.7.1 Relocation Trawling

The intentional capture of ESA-listed species by relocation trawling may be used to assess or reduce the abundance of ESA-listed species in a project location to minimize the risk of lethal encounters with a hopper dredging operation. Modified shrimp trawling equipment is used to sweep the sea floor to either startle ESA-listed species out of the area, with open net relocation trawling, or to capture and often relocate these species, through the use of closed net relocation trawling. This management technique was originally initiated in the early 1980s at Canaveral Harbor, Florida (Rudloe 1981) and has continued to be used as a take minimization measure for dredging in the southeast.

Relocation trawling must maintain a safe distance from the hopper dredge and other vessel traffic in the area. Therefore, the trawler is often not working directly in front of the hopper dredge, but is instead continuously working to remove ESA-listed species from the general dredging area. Trawlers may sometimes need to leave the dredge footprint such as a navigation channel to avoid collision with vessels in the area; however, this will be avoided as much as possible to prevent damage to surrounding habitat. Relocation trawling vessels are also smaller than hopper dredges and therefore more restricted by the weather conditions in which they can safely operate. Relocation trawling will be used as part of the activities proposed, as described and limited by the relocation PDCs in Appendix B. The relocation PDCs include limitations on the time nets are towed and where additional consideration of the use of relocation trawling is necessary such as within the range of ESA-listed corals. Since relocation trawling is typically only used with hopper dredging, relocation trawling in sturgeon rivers will be limited since most work in rivers is completed by other dredging equipment types and additional dredging related PDCs in the Sturgeon PDCs in Appendix E limit the timing of work in these rivers. Appendix H provides additional species-specific guidance on how species captured will be handled including how to safely hold each species, where to release those captured in relocation trawling, when to release animals without bringing them aboard a relocation trawler (e.g., elasmobranchs and leatherback sea turtles), and the order multiple species captured in the same net should be released to minimize harm to them. The decision of when relocation trawling is implemented is described in the risk-based assessment process in Section 2.9 of this Opinion.

2.7.2 ESA-listed Species Handling and Data Collection

ESA-listed species captured on projects covered under this Opinion will be handled by qualified PSOs aboard the vessel who will be responsible for collecting measurements, recording and reporting data, tagging, and taking genetic samples of the captured species. Species specific handling guidelines are provided in the PSO PDCs in Appendix H that detail how the PSO will perform these tasks such as how to take a genetic sample on a specific species, when species should be brought on board or released directly into the water, and how to handle animals in distress, among others.

2.7.3 Monitoring by Aerial Surveys

Aerial surveys are a method used to detect the presence of large marine species, such as North Atlantic right whales, rays, sharks, and large turtles.

Aerial surveys are flown each year as part of the memorandum of agreement between NMFS, USACE, U.S. Navy, and U.S. Coast Guard as part of the Early Warning System and Surveys. Survey are coordinated with a similar aerial survey projects (e.g., the current NMFS-funded Georgia Department of Natural Resources aerial survey project) to optimize joint coverage. North Atlantic right whale observations from aerial surveys are combined with a network of land based volunteer observers are reported to the Early Warning System network. This network alerts mariners in the area of the presence of a right whale generally within 30 minutes of detection alerting them to alter coarse, when possible, to avoid vessel collision with this listed species.

Surveys are conducted according to guidance provided annually by NMFS in coordination with the USACE. For all areas to be surveyed, an aerial survey team experienced in large whale aerial surveys is required to be available to fly 7 days per week; however, priority is given to safety of the aircraft crew and pilots, which may limit the ability to fly every day. Surveys are also only conducted when visibility is sufficient to detect the presence of North Atlantic right whales. As a result, surveys are typically only flown a couple of days a week.

Aerial surveys conducted to detect presence of North Atlantic right whales are detailed in the North Atlantic Right Whale Conservation Plan in Appendix F, and include continuing to fly the aerial survey used as part of the Early Warning System and expanding the area in which surveys will be flown, by adding 2 additional survey teams.

2.8 Action Area

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02). Effects of the action include "all consequences to listed species or critical habitat that are caused by the proposed action," including effects that "may occur later in time and may include consequences occurring outside of the immediate area involving the action" (50 C.F.R. 402.02).

The action area for this programmatic Opinion ("2020 SARBO action area") includes waters off of the Atlantic coast, from the North Carolina/Virginia border south to the tip of Florida including the Florida Keys, and waters off of the Islands of Puerto Rico and the U.S. Virgin Islands (Figure 22). The area evaluated for a specific project covered under this Opinion is limited to the area where direct and indirect effects from that project will occur (the "project action area"). The areas where projects can occur within the broader action area are limited by the PDCs of this Opinion. For projects occurring within the range of ESA-listed corals or Johnson's seagrass, the project action area is defined as including both (1) the project footprint where equipment is located and/or work is occurring and (2) the required survey areas surrounding that work (Appendix C defines those areas within the range of ESA-listed corals and Appendix D defines those areas within the range of Johnson's seagrass).

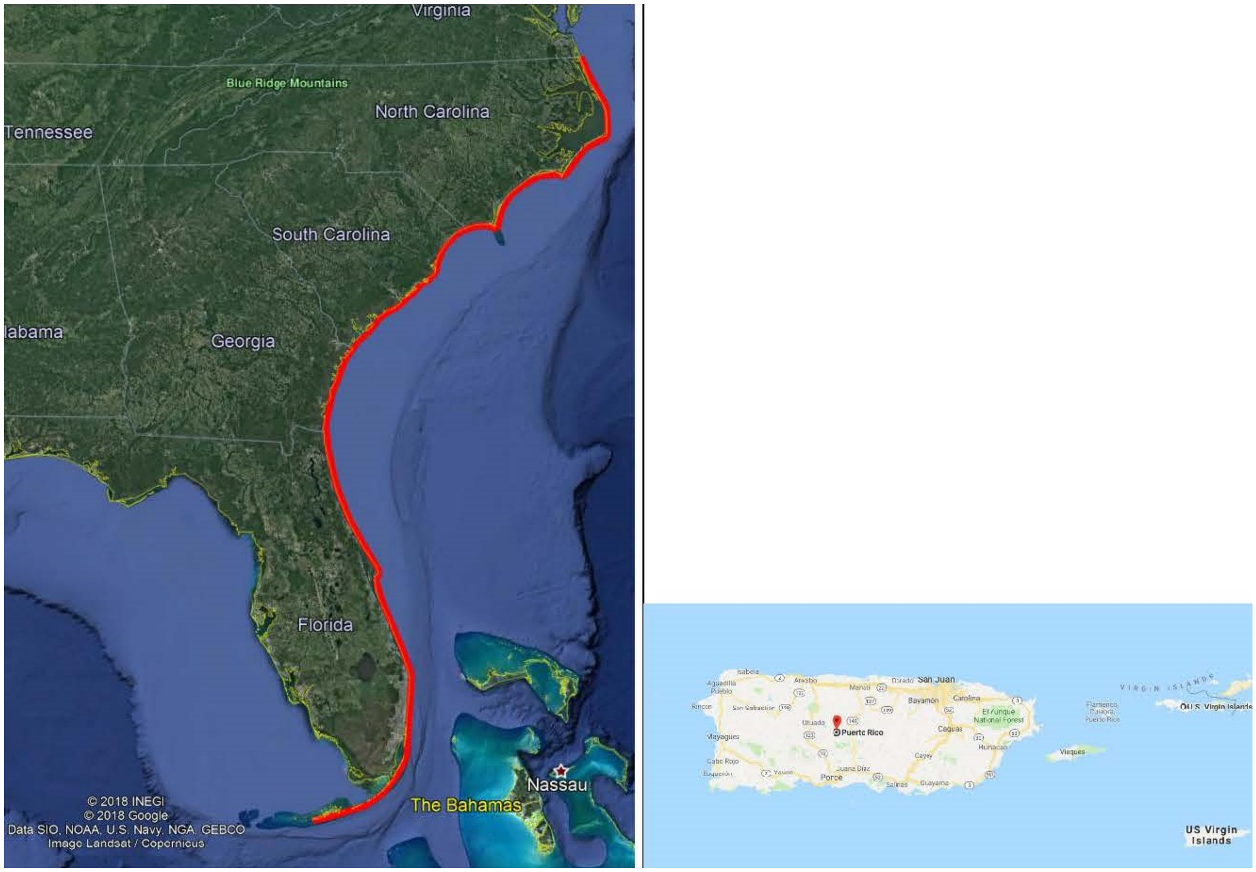

Figure 22. General Action Area for SARBO[16].

The USACE and BOEM have jurisdiction over project locations and activities that meet the PDCs of this Opinion, as described below:

- The USACE has jurisdiction over the activities occurring in state waters within the action area (0-3 nautical miles). Figure 23 shows a map of the USACE jurisdictional boundaries for areas covered under this Opinion, with 4 District offices (SAW, SAC, SAS, and SAJ) operating under the direction of the USACE SAD. Coordination of this Opinion is through the USACE SAD, which oversees the applicability of the 2020 SARBO in the district offices.
- The BOEM has jurisdiction over dredging of sediment from sand lease sites in federal waters (3 – 200 nm) used to nourish beaches coordinated with the USACE. BOEM maintains a website (https://mmis.doi.gov/BOEMMMIS/) that provides up to date data regarding executed leases associated with BOEM's Marine Minerals Program.
- EPA has jurisdiction over designation of ODMDS locations where dredging material may be placed. EPA is not an action agency for purposes of this consultation. EPA and other third party action agencies are encouraged to carefully review this document in making

[16] The red line in the image on the left identifies the states on the Atlantic Ocean that are part of the action area from North Carolina to the Florida Keys. The image on the right shows the U.S. Caribbean also included in the action area including Puerto Rico and the U.S. Virgin Islands.

their determinations regarding consultation requirements under the Endangered Species Act.

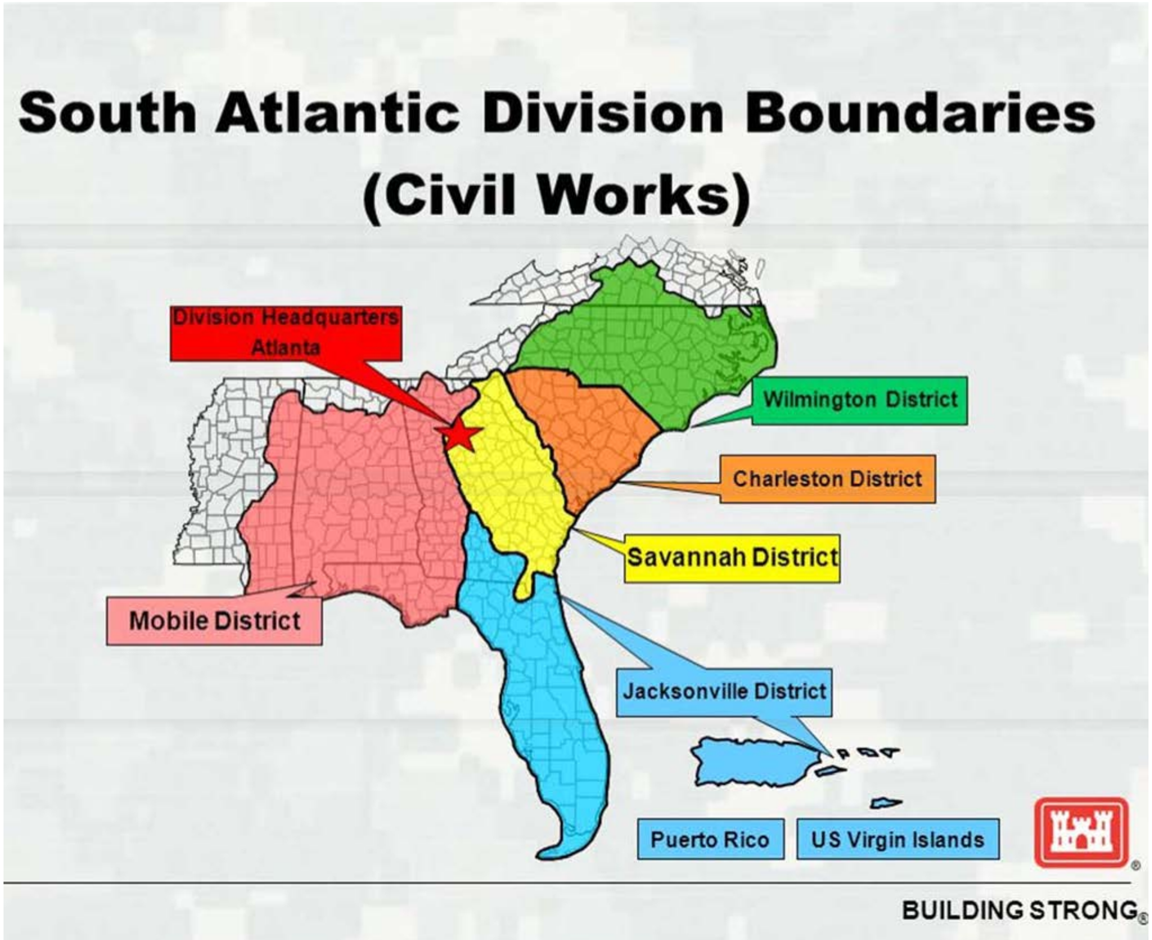

Figure 23. Map of the USACE SAD jurisdictional boundaries.

2.8.1 Navigation Dredging Locations

As needed and as funding is available, USACE Civil Works conducts maintenance dredging in the navigation channels required to be maintained under Title 33 (Navigation and Navigable Waters) listed below. Maintenance dredging other channels and canals used for navigation that are not required to be maintained under Title 33 are also covered under this Opinion, if the original dredging was federally authorized or permitted, occurs in areas covered under this Opinion, and meets the PDCs of this Opinion. Dredging projects outside of navigation channel maintenance are also covered under this Opinion including maintenance dredging in non-federal channels, ports, berths, marinas, boat ramps, and around docks, as described in Section 2.3 of this Opinion. SARBA Appendix B provides detailed descriptions of the routinely maintained navigation channels organized by District. Sections 2.8.1.1 – Section 2.8.1.4 of this Opinion also lists the routinely maintained navigation channels.

Alternative review: Note that in limited instances, projects that deviate from this Opinion's PDCs in a minor way, including location, may be covered under the Supersede process outlined in Section 2.9.5 of this Opinion.

2.8.1.1 USACE SAW Civil Works Maintenance Dredging Projects

The IWW (from Virginia state line to South Carolina state line), Manteo (Shallowbag) Bay, Stumpy Point Bay, Channel to Rodanthe and Rodanthe Harbor, Channel to Avon and Avon Harbor, Swanquarter Harbor, Rollinson Channel, including Channel from Hatteras Inlet to Hatteras, Channel to Silver Lake Harbor (including Big Foot Slough), Ocracoke Inlet, Carteret County Harbors of Refuge, Waterway Connecting Pamlico Sound and Beaufort Harbor, Channel from Back Sound to Lookout Bight, Morehead City Harbor, Beaufort Harbor and Morgan Creek, Atlantic Beach Channels, Peletier Creek, Bogue Inlet, New River Inlet, Channel to Jacksonville, New Topsail Inlet, New Topsail Inlet Connecting Channels, Wrightsville Beach Connecting Channels, Masonboro Inlet, Carolina Beach Inlet, Wilmington Harbor, Cape Fear River above Wilmington, Lockwoods Folly Inlet, Lockwoods Folly River, and Shallotte River.

2.8.1.2 USACE SAC Civil Works Maintenance Dredging Projects

Little River, Murrells Inlet, Georgetown, Jeremy Creek (turning basin inland), Town Creek (McClellanville), Charleston Harbor, Ashley River, Folly River, IWW (from North Carolina state line to Port Royal Sound, South Carolina), and Port Royal.

2.8.1.3 USACE SAS Civil Works Maintenance Dredging Projects

Savannah Harbor, Brunswick Harbor and the IWW (from Port Royal Sound, South Carolina to Cumberland Sound, Georgia).

2.8.1.4 USACE SAJ Civil Works Maintenance Dredging Projects

IWW (from Fernandina Harbor, Florida to Miami, Florida), Okeechobee Waterway from the IWW to the St. Lucie Lock and Dam, Kings Bay Entrance Channel/Inner Channel, Jacksonville Harbor, St. Augustine Harbor, Ponce De Leon Inlet, Canaveral Harbor, Fort Pierce Harbor, St. Lucie Inlet, Palm Beach Harbor, Hillsboro Inlet, Port Everglades, Bakers Haulover Inlet, Miami Harbor, and Key West Harbor in Florida; San Juan Harbor, Arecibo Harbor, Mayagüez Harbor, Ponce Harbor, Yabucoa Harbor, Guavanes Harbor, and Fajardo Harbor in Puerto Rico; St. Thomas Harbor, St. Thomas, and Christiansted Harbor, St. Croix, in the U.S. Virgin Islands.

2.8.2 USACE and BOEM Borrow Area/ Sand Lease Areas

The USACE uses numerous borrow areas including sand acquired during dredging, especially in inlets and passes. Borrow areas can also be offshore in state waters under the jurisdiction of the USACE or in federal waters under the jurisdiction of BOEM. The borrow areas used for specific beach nourishment projects are identified in SARBA Appendix B. As stated earlier, the BOEM website (https://mmis.doi.gov/BOEMMMIS/) provides up to date data regarding executed leases associated with BOEM's Marine Minerals Program including borrow sites covered under this

Opinion (Table 5). New borrow areas are also allowed under the 2020 SARBO if they meet the requirements of the PDCs.

Alternative review: Note that in limited instances, projects that deviate from this Opinion's PDCs in a minor way, including location, may be covered under the Supersede process outlined in Section 2.9.5 of this Opinion.

Table 5. BOEM Borrow Sites

Lease Status	State	Lease Number	Project Name	Borrow Area	Total Sand Allocated (cy)	Effective Date	Expiration Date	Size (acres)
Active	North Carolina	OCS-A-0523	Carteret County 2019 - Bogue Banks	Morehead City ODMDS	2,000,000	2/21/2019	2/21/2021	218
Active	South Carolina	OCS-A-0477	Port Charleston 2010	East Excavation, West Excavation	6,000,000	3/23/2010	7/20/2019	808
Active	South Carolina	OCS-A-0514	Myrtle Beach 2016	Surfside	1,600,000	9/12/2016	10/17/2020	426
Proposed	Florida		Flagler County (Proposed 2014)					4,845
Proposed	Florida		Miami-Dade County (Proposed 2015)		5,200,000			2,614
Proposed	Florida		St. Johns County (Proposed 2016)					13,419
Proposed	Florida		St. Lucie County (Proposed 2012)					170
Proposed	Florida		Brevard County (Proposed 2016) - Mid Reach		900,000			1,469
Proposed	South Carolina		Folly Beach (Proposed 2016)	Borrow Area C, Borrow Area D				142

2.8.3 ODMDS

U.S. Environmental Protection Agency ODMDS sites currently federally-authorized include the list below. A list of the ODMDS locations is included in Table 6 below. This list includes the current ODMDS sites within the action area as of July 2019, according to the EPA website (https://www.epa.gov/ocean-dumping/ocean-disposal-map). Only designated ODMDS locations are covered under this Opinion. Therefore, ODMDS locations authorized after the completion of 2020 SARBO may be covered by this Opinion if a separate Section 7 consultation has been completed or through the Supersede review process outlined in Section 2.9.5 of this Opinion.

Alternative review: Note that in limited instances, projects that deviate from this Opinion's PDCs in a minor way, including location, may be covered under the Supersede process outlined in Section 2.9.5 of this Opinion.

Table 6. EPA ODMDS Sites

Site Name	Location	EPA Region	Date of Site Designation	Site Area (nmi^2)	Average Site Depth (ft)
Canaveral Harbor	Florida	4	10/22/1990	4	51
Fernandina Beach	Florida	4	3/25/1987	4	52
Fort Pierce Harbor	Florida	4	10/4/1993	1	47
Miami	Florida	4	2/29/1996	1	607
Palm Beach Harbor	Florida	4	2/17/2005	1	575
Port Everglades Harbor	Florida	4	2/17/2005	1	673
Jacksonville	Florida	4	7/5/1984	4.56	45
Brunswick Harbor	Georgia	4	1/23/1989	2	30
Savannah	Georgia	4	8/3/1987	4.26	37
Morehead City	North Carolina	4	9/14/1987	8	39
New Wilmington	North Carolina	4	8/5/2002	9.4	44
Wilmington	North Carolina	4	8/3/1987	2.3	43
Arecibo Harbor	Puerto Rico	2	10/20/1988	1	850
Mayaguez Harbor	Puerto Rico	2	10/20/1988	1	1,206
Ponce Harbor	Puerto Rico	2	10/20/1988	1	1,289
San Juan Harbor	Puerto Rico	2	4/22/1988	0.98	958
Yabucoa Harbor	Puerto Rico	2	10/20/1988	1	2,400
Charleston	South Carolina	4	8/3/1987	7.4	38
Georgetown Harbor	South Carolina	4	10/27/1988	1	28
Port Royal	South Carolina	4	10/24/2005	1	36

2.8.4 Beach Nourishment Locations

Beach nourishment projects covered under the 2020 SARBO are limited to nourishment in areas defined in SARBA Appendix B and new locations outside of the range of ESA-listed corals that meet the PDCs of this Opinion.

Current federal Coastal Storm Risk Management (i.e., beach nourishment) and/or ecosystem restoration projects are included in the list below. SARBA Appendix B provides a detailed description of current Coastal Storm Risk Management projects as well as other federally permitted beach nourishment projects.

- <u>Wilmington District Projects</u>: Carolina Beach, Kure Beach, Ocean Isle, Wrightsville Beach
- <u>Charleston District Projects</u>: Edisto Beach, Folly Beach, Myrtle Beach, Pawleys Island, Hunting Island
- <u>Savannah District Projects</u>: Tybee Island

- Jacksonville District Projects: Brevard County, Broward County, Dade County, Duval County, Flagler County, Indian River County, Martin County, Miami-Dade, Nassau County, Palm Beach County (Lake Worth Inlet to South Lake Worth Inlet), Palm Beach County (Martin County Line to Lake Worth Inlet & South Lake Worth Inlet to Broward County Line), St. Johns County, Volusia County, St. Lucie County.

Alternative review: Note that in limited instances, projects that deviate from this Opinion's PDCs in a minor way, including location, may be covered under the Supersede process outlined in Section 2.9.5 of this Opinion.

2.8.5 Pipeline Corridors

The USACE uses designated pipeline corridors to place sand for beach nourishment projects. Previously designated and/or used corridors for particular projects are defined in SARBA Appendix B by beach location. New pipeline locations are also allowed under this Opinion outside of the range of ESA-listed corals. Existing and new pipeline corridors are limited by the PDCs in this Opinion, with increased protections provided for existing corridors within the range of ESA-listed corals.

Alternative review: Note that in limited instances, projects that deviate from this Opinion's PDCs in a minor way, including location, may be covered under the Supersede process outlined in Section 2.9.5 of this Opinion.

2.9 Programmatic Implementation, Tracking, and Reporting

This section outlines the process used to determine if covering a project under this Opinion is appropriate; how projects covered by this Opinion are reported and tracked; how coordination will continue between NMFS, USACE, and BOEM; and how the aggregate effects of all activities occurring during a year will be reviewed to determine if the effects of the proposed activities, including the level of take or loss of critical habitat, exceeded the amount analyzed in this Opinion. Most of the processes outlined in this section are a continuation of how coordination between the USACE and NMFS has been handled for many years, with some modifications and additional specific reporting requirements. The USACE has successfully managed its dredging program throughout the Southeast for many decades in a manner that minimizes impacts to ESA-listed species and critical habitat. Protective measures used by USACE have included evaluating the risk to ESA-listed species that may be associated with construction of a specific project prior to construction, applying risk minimization measures when and where they were deemed appropriate, re-evaluating the risk to ESA-listed species following each take, tracking all take in real time (first on the Sea Turtle Data Warehouse, now reported in the Operations and Dredging Endangered Species System [ODESS][17]), and continuing to coordinate with NMFS when questions or issues arise. The addition of BOEM to this consultation is limited to dredging activities in federal waters and the inter-connected material placement activities, as described in Section 2.2 of the Opinion. As part of the SARBO

[17] https://dqm.usace.army.mil/odess/#/home

Team (consisting of members of the USACE, BOEM, and NMFS), the USACE and BOEM helped develop and have agreed to all of the measures of this Opinion including the PDCs provided in Appendix A-H and the processes and requirements outlined in Section 2.9 of this Opinion that formalizes how all projects covered under this Opinion will be implemented, tracked, and reported.

2.9.1 USACE and/or BOEM Project-Specific Review for a Project to be Covered under SARBO

Before the USACE and/or BOEM authorize an activity covered under this Opinion, they must conduct a project-specific review of all project details to ensure compliance with all applicable PDCs. If the PDCs are met, then the project qualifies for coverage under this Opinion. If staff determines that some portion of the project would require a deviation from the PDCs, but the project would still have substantially similar effects to those considered in this Opinion, USACE/BOEM will contact NMFS to determine if coverage would be appropriate under the 2020 SARBO alternative review processes outlined in Section 2.9.5 of this Opinion (Alternative Project Implementation and Programmatic Modification through the Superseding Process of Review).

Projects that can be covered under this Opinion may be authorized either individually by the USACE or BOEM or may be authorized jointly if a project includes components occurring in both state and federal waters. The federal authorization process by these agencies is briefly outlined below:

- For projects authorized solely by the USACE: This Opinion was developed in coordination with the USACE SAD and will be overseen at the Division level for both Civil Works and Regulatory projects. If the USACE Districts within the action area (SAW, SAC, SAS, and SAJ) have questions about the adherence to PDCs or reporting requirements, they will contact the USACE SAD Office point of contact (as designated by the USACE SAD Operations and Regulatory Division Chief) for guidance and coordination with NMFS. While projects may be evaluated for coverage under this Opinion at the USACE District level, the USACE SAD remains responsible for all compliance and reporting requirements. As stated under Programmatic Review (Section 2.9.4 of this Opinion), the USACE SAD (Operations) will conduct an internal review of the projects authorized under SARBO each year to ensure that all USACE project managers permitting projects covered under this Opinion are applying SARBO appropriately.

- For projects authorized solely by the BOEM: In rare instances, BOEM may authorize dredging/ sand mining in federal waters for a project that does not require USACE coordination (e.g., if the sand is placed on a beach above mean high water). In those instances, BOEM will be independently responsible for all implementation and reporting requirements of the 2020 SARBO and notify the USACE in advance of the application of the SARBO. BOEM will send completed reports to the USACE SAD for inclusion in the SARBO annual review and report, as stated under Programmatic Review (Section 2.9.4 of this Opinion).

- For projects jointly authorized by the USACE and BOEM: For projects where portions of the project occur within both USACE and BOEM's jurisdiction, the USACE and BOEM will determine the appropriate lead action agency for the project. The lead action agency will be responsible for ensuring work is carried out in accordance with this Opinion in coordination with the other agency. As described above, the USACE SAD remains the primary point of contact within the USACE for questions of applicability of this Opinion to a project. All reports completed will be provided to the USACE SAD according to the tracking and reporting requirements outlined in Section 2.9.3 of this Opinion.

- For projects jointly authorized with another Federal agency: There may be projects where USACE or BOEM is an action agency, the proposed project is in the action area and consistent with the 2020 SARBO PDCs, and portions of the project are coordinated with or completed by another Federal agency (e.g., USACE authorizing maintenance dredging by the U.S. Air Force in waters off of a military base, or EPA authorizing USACE use of an ODMDS). These partner agencies are encouraged to carefully review this document in making their determinations regarding their consultation requirements under the Endangered Species Act and to consider whether their action results in any additional or different effects not addressed herein. In the event that such a late-arriving partner action agency requests consultation on an action consistent with the PDCs of this Opinion in the action area, NMFS will evaluate what further steps may be required to meet the requirements the Endangered Species Act.

2.9.2 Risk-based Adaptive Project Management

The proposed action as analyzed in this Opinion allows some flexibility in the timing of project completion through the use of a risk assessment and risk management process, outlined below in Section 2.9.2.2 of this Opinion. Using this risk-based decision-making process, dredging will be allowed outside of the previously established seasonal dredging windows required in the 1997 SARBO.

2.9.2.1 History of Adaptive Management

Under the 1997 SARBO, the USACE retained flexibility, within defined seasonal dredging windows, to decide when and where projects would occur and the equipment type used for a particular project. The USACE SAD developed a *Risk Assessment and Risk Management Plan*[18] (Plan), to help guide their decision-making process and to address circumstances which may have contributed to an incidental take. The Plan included documenting how required hopper dredging conditions were met, including the use of measures like turtle deflecting dragheads, proper inflow and overflow screening and ensuring that dredge pumps were disengaged when the draghead was not firmly planted in the sediment. This Plan also outlined procedures to follow when take occurred to reduce the risk of further take. The USACE's practice has been to update this Plan annually to minimize risk to ESA-listed species.

Utilizing adaptive management in this manner allowed the USACE to consider the anticipated risk of harm to ESA-listed species in the context of shifting variables (e.g., environmental, financial, regulatory, etc.). Subsequent decisions made regarding project timing and equipment use maximized the ability to complete dredging and material placement projects, while minimizing the risk of incidental take. The USACE has a proven history of using this process to

[18] https://dqm.usace.army.mil/odess/documents/GuidelinesRiskAssessmentsRiskMgmtPlans.pdf

further reduce the likelihood of incidental take, and will continue to do so under the 2020 SARBO.

2.9.2.2 SARBO Risk Assessment and Risk Management Process

This Opinion formalizes and expands the risk assessment process to include BOEM and to include coordination and input from the SARBO Team (consisting of members of the USACE, BOEM, and NMFS), throughout the life of this Opinion (e.g., until reinitiation triggers are met as outlined in Section 12 of this Opinion).

The risk-based adaptive project-management process involves the consideration of institutional knowledge of particular project sites, the potential effects to ESA-listed species and designated critical habitat, and the use of any current or new best available information. The SARBO team will coordinate as appropriate to discuss project specific issues and meet at least once annually to discuss the projects proposed to be covered under this Opinion for the upcoming year and associated minimization measures that may be considered to reduce take for particular projects.

The SARBO Risk Assessment and Risk Management Process will consider the steps outlined below and detailed in Appendix J. Each step outlines the general process used to evaluate risk from projects and how minimization measures will be selected to reduce the risk of lethal take. This process will continue to be refined by the SARBO Team as this Programmatic Opinion is implemented.

- Assessment Step 1. Determine the list of upcoming projects expected and pre-construction risk assessment

  Each fiscal year, the USACE and/or BOEM will compile a list of projects proposed for the next year and beyond (e.g., projects proposed for the next 1-5 years), including relevant minimization measures based on the pre-construction risk assessment results. The final project timing and risk assessment will be developed and maintained by the USACE and/or BOEM. Timing of upcoming projects will minimize the risk of impacts to ESA-listed species by considering the risk to ESA-listed species posed by particular projects based on project-specific timing, location, and equipment used, as appropriate. This assessment will involve considering the presence of ESA-listed species at project locations/times, known equipment interactions with species expected to be present, and the history of interactions at a particular project site.[19] These suggested minimization measures consider when, where, and what equipment could be used to reduce take based species use of an area. This information combined with past experience by the USACE and BOEM of problems encountered working in the same or similar areas will continue to be incorporated into the pre-construction risk assessment.

  Individual projects that were not reviewed during the annual review (e.g., USACE Regulatory project that are proposed after the annual review and will be implemented before the next annual review) will be reviewed using the same approach and discussion with NMFS. Before permitting any activities analyzed under the 2020 SARBO, conformance with the PDCs in the 2020 SARBO must be confirmed.

[19] NMFS has provided an initial list of specific suggested items to consider when determining how to reduce take from an upcoming project (Appendix J); however, the project-specific considerations used are expected to evolve for each equipment type and project area, as USACE and BOEM continue to engage in projects in the action area.

- Assessment Step 2. Post-take Risk Assessment
  This process will be completed by USACE and/or BOEM after any take occurs to determine what factors lead to the adverse effect and if additional measures can be used to prevent it from occurring again.
- Assessment Step 3. Post-Project Review and Reporting
  This process will be used to document what happened during the project and any lessons learned that can be applied to future projects to reduce the risk of incidental take.
- Assessment Step 4. Annual Review and Reporting
  This process will be used to document what happened during the year and any lessons learned that can be applied to future projects to reduce the risk of incidental take.

2.9.3 SARBO Team Communication and Reporting

The USACE in coordination with BOEM will inform NMFS of upcoming projects covered under this Opinion, track, and report issues that arise during construction, and track and report relevant details of the project evaluated under this Opinion. NOTE – Reporting is not just for Hopper dredging – information from all types of dredging and placement covered under this Opinion must be reported.

2.9.3.1 Digital Reporting

All reporting requirements of this Opinion will be provided digitally (e.g., ODESS, emailed to NMFS at **SERO.Dredge@noaa.gov**, or using any other communication method agreed upon by the SARBO Team) in a format that NMFS can review, edit, sort by the information categories listed below, and manipulate the data for the purposes of calculations and comparisons (e.g., digital spreadsheet or compatible format, not a scanned pdf). Currently, ODESS is the primary reporting system used by the USACE to store and monitor dredging project data including information associated with encounters with ESA-listed species. If the USACE in coordination with BOEM develop a different web-based tracking system or update ODESS, the reporting requirements outlined in Section 2.9 of this Opinion will still apply and the system will be accessible to NMFS staff for review and downloading of all required information. ODESS will continue to be developed as needed, based on funding availability, to support priority data needs identified by the SARBO team. NMFS encourages the use of public data sharing formats such as ODESS to the maximum extent possible so that necessary information is accessible in one location for monitoring convenience by the SARBO Team, concerned public citizens, and entities that must be provided information as a condition of this Opinion (e.g., state sea turtle coordinator notifications required by PDC REPORT.1). If a public website like ODESS is used to meet the reporting requirements, a notification still needs to be sent to **SERO.Dredge@noaa.gov** alerting NMFS that take has occurred or that new project information has been added to the website.

2.9.3.2 Meetings

The SARBO Team will conduct at least 1 annual meeting as part of the programmatic annual review described in Section 2.9.4 of this Opinion and will conduct calls throughout the year as needed to discuss how this Opinion is being implemented. It is anticipated that quarterly calls may be needed to discuss issues as they arise or to specifically address other reporting requirements including (1) pre-construction notification as part of a kick-off meeting for the dredging year, risk-based assessment concerns, or the annual review. However, the number and frequency of the meetings may be adjusted by the SARBO Team.

2.9.3.3 Pre-Construction Notification:

NMFS will be notified at least 2 weeks prior to construction of any project covered under this Opinion by the USACE and/or BOEM so that NMFS is aware of current and upcoming projects in the region. The notification will include the required project information provided in Section 2.9.3.5 of this Opinion that explains what the project is, where it will be happening, how it will be completed, and when work is expected to occur. All information will be reported according to the digital reporting requirements provided in Section 2.9.3.1 of this Opinion. The pre-construction notification will be provided in a manner that creates a searchable compiled list of all projects planned to begin within the fiscal year, which could be transmitted by emailing a spreadsheet that is updated with each new project, a list maintained on a publicly available website such as ODESS, or other method approved by the SARBO Team. The pre-construction notification (sent to **SERO.Dredge@noaa.gov**) will include a statement that the applicable PDCs have been reviewed and will be requirements of the project.

2.9.3.4 During and Post-Construction Reporting

Important project details will be reported to NMFS digitally, according to the digital reporting requirements provided in Section 2.9.3.1 of this Opinion. This includes:

- All lethal and nonlethal take associated with a project covered under this Opinion will be reported within 48 hours. Project details related to take that will be reported as detailed in Section 2.9.3.5.2 of this Opinion.
- All observations of North Atlantic right whales observed while completing a project (aerial survey reporting is outlined separately in Appendix F) be reported within 24 hours of the observation. The process to report a North Atlantic right whale observation is outlined in the North Atlantic Right Whale Plan (Appendix F) and applies to all work covered under this Opinion.
- Any reporting requirements outlined in the PDCs including surveys conducted under the Coral PDCs (Appendix C), surveys conducted under the Johnson's seagrass PDCs (Appendix D), and PSO responsibilities outlined in Appendix H.

The SARBO Team must be able to access and track relevant project details to verify compliance with the PDCs of this Opinion including the ability to monitor the accumulating total take of ESA-listed species and any loss of designated critical habitat features for the year, though loss of critical habitat is not anticipated. Project details that will be reported for all projects (regardless of if take occurred) are detailed in Section 2.9.3.5.1 of this Opinion.

2.9.3.5 Required Project Information:

Project details listed below apply to all projects covered under this Opinion, even if the project did not include hopper dredging, resulted in no take of an ESA-listed species, or resulted in no adverse effects to critical habitat. All required information will be digitally accessible to NMFS prior to work commencing and reported according to the digital reporting requirements provided in Section 2.9.3.1 of this Opinion. Information initially provided as estimated project details, such as the start date and the total volume of material dredged, will be updated with accurate final information and digitally available to NMFS within 30 days of project completion.

This information required is intended to provide the basic details that were needed to complete the analysis in this Opinion and are needed to confirm that the effects evaluated in this Opinion are still accurate. These details will be reviewed during the programmatic annual review (Section 2.9.4.1 of this Opinion), may be incorporated in the risk-based adaptive management process for future projects occurring in the general area of a completed project (Section 2.9.2.2 of this Opinion), and may be used to inform future consultations on similar actions analyzed in this Opinion.

2.9.3.5.1 Required Project Information for All Projects

The required project details listed below are grouped by the questions they answer with an explanation of why the reported information is important to the implementation of the 2020 SARBO and future similar consultations.

Who is in Charge of the Project?
It is important to track which action agency (e.g., USACE or BOEM) and point of contact is overseeing the project and if another action agency involved. Knowing who is in charge of the project and how the project was authorized (e.g., request for SARBO Supersede review for a modification) is important for project tracking and consistency under this Opinion, and if there are questions later about the rationale behind decisions made. If the project includes a PSO, the PSO and PSO company name and contact information is important if there are questions about take. The following information will be provided to NMFS:

1. USACE and/or BOEM Project Manager (point of contact and contact information)
2. Protected Species Observer/s: Observer company, if a PSO was used, and contact information
3. Each federal action agency associated with project (e.g., USACE SAD, SAW, SAC, SAS, SAJ, BOEM, other agency consultation on the project such as the U.S. Air Force and/or Federal Emergency Management Agency [FEMA])
4. All federal action agency project tracking numbers associated with the project, if applicable (e.g., USACE Regulatory tracking number, e.g., SAW-2018-xxxxx)

5. Biological Opinion(s) used to authorize the work (i.e., SARBO and any other Opinion used to cover a proposed project, if combined)

When is the Project Occurring?
The estimated start and end date will be provided in the pre-construction notification (Section 2.9.3.3 of this Opinion) and then updated to the actual start and end date. Knowing when a project occurs is important in understanding the risk of the activity to ESA-listed species since it may or may not be present in the area when work is proposed or may be using the area for a specific life function in that location during that time of year, such as the presence of the North Atlantic right whale during calving season. The following information will be provided to NMFS:

1. Project start date (Estimated dates must be updated with actual dates)
2. Project end date (Estimated dates must be updated with actual dates)

Where is the Project Occurring?
Knowing the project overall location and the specific area where within the project area where work occurred is important to be able to determine how the project spatially relates to other factors. This could include being able to overlay how many projects occurred in a critical habitat unit or an area that required additional PDCs (e.g., within the range of ESA-listed corals) to see if the effects analyzed in this Opinion are accurate. Tracking which projects are occurring in sensitive areas is important to ensuring the effects analyzed in this Opinion are accurate. Knowing where a project occurs could also be used to determine if reported strandings in an area could be linked to work occurring under this Opinion.

If the extent of the project footprint (e.g., the entire extent of ABC Borrow area) has already been provided to NMFS or is available for download from a specified public website, referring to the location in a manner that is quantifiable is sufficient (XYZ Beach from mile marker X-Y). If it is a new location, the geographic limits of the project footprint need to be provided as a shapefile. The following information will be provided to NMFS:

1. Project name (Typically projects are referred to by the name of the area. If the area has more than one common name, all common names should be provided).
2. Project location for both dredging AND placement. For regularly occurring projects with an easily referenced named location, a central location may be sufficient (e.g., latitude and longitude in decimal degree format [xx.xxxx, -xx.xxxx]). Project spatiolocation (i.e., shapefile/Keyhole Markup language Zipped (commonly referred to as KMZ)/ geographic information system (commonly referred to as GIS) layer to show the complete action area is needed if this information has not been previously provided to NMFS such as a USACE regulatory project that provided during the completion of this Opinion or the area of a channel realignment covered under this Opinion.
3. Is the project occurring in an area identified in this Opinion that requires additional protection, such as within the range of ESA-listed coral (Appendix C), Johnson’s seagrass (Appendix D), sturgeon rivers (Appendix E), or when and where North Atlantic right whales may be present (Appendix F)?

4. Is the project occurring within the geographic limits of a designated critical habitat, even if features are not impacted? For example, Johnson's seagrass critical habitat Unit J or loggerhead critical habitat unit LOGG-N-19.
5. Total area of the project that occurs within the geographic area of one or more critical habitat units, if applicable. For example, 1,000 ft² of dredging occurred within North Atlantic right whale critical habitat.

What Type of Project and Equipment?

In order to track if the effects analyzed in this Opinion are accurate and to know if the number of each species estimated to be captured based on the amount of anticipated dredging estimated to occur annually under this Opinion (catch per unit effort [CPUE]) is accurate, tracking the types of projects covered under this Opinion and the types of equipment used is needed.

This information may start to show trends that can be used for future projects and/or future dredging consultations to reduce take of ESA-listed species. One example would be if take is reduced when bed-leveling is used during the clean-up phase of hopper dredging in most locations, but not in certain other locations or for specific bed-leveling designs, this information could be investigated further and used in future risk-based assessments regarding the type of equipment that could be used in a specific location to reduce take. The following information will be provided to NMFS:

1. Project type/s
   a) Maintenance Dredging
   b) Minor channel modification/realignment
   c) Borrow site
   d) Muck dredging
   e) Beach nourishment
   f) Nearshore placement
   g) ODMDS
   h) G&G survey
   i) New placement location
   j) Other
2. Pre-project proposed dredge and placement total volume in cubic yards.
3. Post-project actual dredge and placement total volume in cubic yards.
4. Confirmation (yes/no) that dredging does not exceed the previously federally-approved or federally-authorized dredge template including previously considered overdepth and/or advanced maintenance. If it does exceed (yes), an explanation will be provided (e.g., approved through supersede, unintentional/unusual event and lesson learned).
5. Vessels and specific equipment used on project. A single project may include more than 1 category of equipment listed below for a portion or all of a project. The equipment types

expected to be used and listed with the pre-construction notification (Section 2.9.3.3 of this Opinion) will be updated at the end of the project if modifications were necessary.

a) Hopper dredge

  (1) Used UXO/MEC screening. Note that projects that the use of UXO/MEC screening is only allowed if reviewed through the Alternative review/ Supersede process outlined in Section 2.9.5 of this Opinion.

  (2) Screening size used for the project. If the project required an increase or removal of inflow screen size (according to PDC HOPPER.1, Appendix B), the sizes used and volume dredged with screens larger than 4 x 4-inch must be recorded and reported.

  (3) If inflow screening is removed, the USACE and/or BOEM will track the start and end date of dredging that occurred without inflow screening and the number of loads, which will be reported in the annual report.

  (4) Bycatch captured

b) Modified hopper (as defined in Section 2.5.2.2 of the Opinion such as the CURRITUCK and MURDEN).

c) Non-hopper dredging equipment (e.g., bucket, clamshell, cutterhead, water-injection, bed-leveling to complete project)

d) Bed-leveling (used as the sole form of material movement or just during clean-up phase of hopper dredging).

e) Name and automatic identification system tracking number of any support vessels over 33-ft in length in areas and during times that required adherence to the North Atlantic Right Whale Conservation Plan (Appendix F).

f) Geophysical survey

  (1) Include the equipment type (e.g., multibeam, boomer), frequency at which the equipment was operated, maximum source/power level it was operated at (that will be used during the annual review to determine the dB limits in the PDCs were not exceeded), location used, and total time used.

g) Relocation trawling

  (1) Total number of tows for the project.

  (2) Total number of days.

  (3) Relocation trawling start date.

  (4) Relocation trawling end date.

  (5) Bycatch captured (i.e., other species captured during trawling by species and estimated number of captures).

h) New Equipment or construction method approved through the SARBO Supersede 2 process outlined in Section 2.9.5.2 of this Opinion.

2.9.3.5.2 Required Project Information When Take Occurs

The following details will be reported when take occurs associated with a project covered under this Opinion. This required information applies to lethal and nonlethal take of mobile species (i.e., all species listed in Table 8 of this Opinion, except ESA-listed corals and Johnson's seagrass). Information collected provides details on the type of species captured including the size and age of the animal based on the measurements taken. Environmental conditions recorded at the time of take (e.g., Beaufort state, water and air temperature, and notes provided in the comments section) may help to better understand where and when take may occur at future similar projects and may be incorporated into the risk-assessment process. For example, the number of sea turtle takes increases when the water temperature is above or below a certain threshold and after a major cold snap. Tracking this information aids in the risk assessments for future projects. Knowing the Beaufort state also helps to understand how visible animals may be in the area, especially if a vessel strike occurs. The following information will be provided to NMFS:

1. Location of take (latitude and longitude if possible or estimated based on the portion of project where work is occurring such as a specific portion of an entrance channel, pass, or borrow site)
2. Tow number when take occurred during relocation trawling or dredge load number if take occurred during hopper dredging.
3. Protected Species Observer/s that observed and handled the take: Observer name/company and contact information
4. Species take must be tracked by total number (e.g., 3 loggerhead sea turtles). Atlantic sturgeon must be reported by District Population Segment (DPS). Project take details can initially state Atlantic sturgeon DPS unknown, but must be updated to known DPS when the genetic sample is processed, which will occur within 1 year of take (Appendix H). All samples must be processed in time to provide DPS information in the annual report. If the observed remains of a sea turtle cannot be identified by species, recording the take as unknown sea turtle is appropriate. Unknown sturgeon will require genetic testing to determine if it was an identifiable DPS of Atlantic sturgeon.
5. Previous animal identification/tracking tag information (internal and external tags), if any
6. New passive integrated transponder (PIT) Tag information, if inserted according to the PSO PDCs in Appendix H
7. Genetic sample collected, if applicable under PSO PDCs in Appendix H
8. Age class of species take based on size (e.g., juvenile, adult)
9. Specimen Condition (e.g., alive, fresh dead, or decomposed as described in the PSO PDCs in 0 Section 4). While decomposed animals are not counted as take associated with the project, they will still be recorded and reported with the project take.
10. Final disposition (e.g., released at site, relocated, rehabilitation and outcome once known, necropsy, disposal)
11. Species gender (if known)

12. Species size/length (measurement details are provided by species in the PSO PDCs, in Appendix H).
13. Beaufort state at the time of take.
14. Water temperature at the time of take-recorded at the water's surface in marine environments and at the bottom in estuarine and riverine environments.
15. Notes about species condition: Any additional relevant information regarding take of ESA-listed species including turtles with Fibropapillomatosis disease, previous wounds, or multiple ESA-listed species captured in same net.
16. Notes about site condition anomalies: Any observations by PSO or crew that may lead to increased captures or deposition of capture including presence of other species like cannonball jelly fish or regional conditions such as large storm or dramatic change in temperature like a recent cold snap.
17. If the take occurred during hopper dredging:

a) List the location where take was identified (e.g., draghead, inflow box, overflow box).

b) Provide the screening in place at the time of take. Were both inflow and overflow screening used? List the size of screening used for both.

c) State if UXO/MEC screening was installed at time of take

2.9.4 Annual Programmatic Review

The USACE SAD in coordination with BOEM is responsible for implementation, management, and administration of the proposed action for all projects covered under this Opinion and the SARBO Team will continue to coordinate and communicate for the life of this Opinion to ensure its success. This includes an annual review of all activities covered under this Opinion as described in this section. The annual programmatic review will include the discussions outlined in this Section that may be combined or expanded, as deemed appropriate by the SARBO Team.

The annual programmatic review assesses all projects covered under this Opinion to ensure that the Opinion is being implemented in the manner intended and that the effects considered in this Opinion are still accurate. In order to complete the programmatic annual review, the USACE SAD will need to report the necessary details for each project outlined in Section 2.9.3.5 of this Opinion and ensure that the information is accurate and complete. The combined information for all projects covered under the Opinion from the previous fiscal year will then be used to determine if the use of this Opinion to cover a project was appropriate. From NMFS's experience working with other Programmatic Biological Opinions in this region, we suggest that the following items be checked before submitting the programmatic annual review to NMFS:

- Randomly select and review projects covered under this Opinion by staff other than those on the SARBO Team to confirm compliance with the requirements of this Opinion including all applicable PDCs.
- Map all project locations to determine how many occurred in critical habitat.

- Map all project locations to determine how many occurred in areas that required additional PDCs such as those within the range of ESA-listed corals and ensure the additional protective measures were followed.
- Review the compiled spreadsheet to ensure that all information is reported. Certain details may be provided as an estimate during the pre-construction notification and then will need to be updated once work is complete such as the total dredge volume or start and end date.

2.9.4.1 Annual Programmatic Report

Each fiscal year, the USACE SAD in coordination with BOEM will provide an annual programmatic report that includes all information outlined below. The annual review will be completed and provided to NMFS as soon as possible after the end of the fiscal year. The first annual review for the 2020 SARBO implementation will determine how soon an annual review can be accurately and reasonably completed. In addition, the USACE SAD will host an annual meeting for the SARBO Team to discuss the annual report. The SARBO Team will be provided the annual programmatic report with sufficient time to review the information prior to the annual programmatic review meeting. The SARBO Team will work to resolve any outstanding questions or concerns and document the results of these discussions.

The reporting requirements in this section are meant to ensure that this Opinion is protective of ESA-listed species. These requirements may be adapted by agreement between NMFS, USACE, and BOEM, as this Opinion is implemented, in order to ensure accuracy, validity, and utility of data collected and to ensure protection of the species discussed in the Opinion. This will allow maximum flexibility and protectiveness of the species while ensuring operations occur in the safest manner.

Following the annual review, the SARBO Team may jointly determine that revisions to the Opinion or the PDCs may be necessary. If the SARBO team believes that PDCs require minor modification or correction, the process established below for changing PDCs may be initiated (Section 2.9.5.3 of this Opinion). If revisions exceed those that are deemed appropriate for minor modifications to the PDCs, re-initiation of consultation may be required as appropriate as provided in 50 CFR Section 402.16, and detailed further in Section 12 of this Opinion.

The annual programmatic report must include all of the information identified in Section 2.9.4.2 of this Opinion, if relevant to the project covered under this Opinion.

2.9.4.2 Data Required for the Programmatic Annual Review Report

The following information will be reported in a digital compiled and sortable summary spreadsheet or narrative, as appropriate, according to the reporting guidelines provided in Section 2.9.3.1 of this Opinion.

1) This report will include a master spreadsheet compiling all of the required information from Section 2.9.3.5 of this Opinion, for all projects covered by this Opinion during the year. The spreadsheet must provide a tally of at least the number of nonlethal and lethal take by species/DPS, any loss of critical habitat features by critical habitat unit and quantifying any

loss of each feature by the area of loss (acres or square feet),[20] and total volume dredged during the year.

2) In addition to, or as part of, the master spreadsheet identified in item 1 above, identify and tally all projects:

   a) Located within a critical habitat unit or species-specific range that required additional protection, as appropriate:

      i) In sturgeon rivers (Sturgeon PDCs, Appendix E)

      ii) In the range of Johnson's seagrass (Johnson's seagrass PDCs, Appendix D)

      iii) In the range of ESA-listed corals (Coral PDCs, Appendix C)

      iv) In the range and during the time when North Atlantic right whales may be present (Appendix F)

   b) Using an equipment type that required additional reporting, such as:

      i) geophysical surveys

3) Hopper dredging with modified or removed inflow screening.

4) Project activities located within the range of ESA-listed corals that required a survey. Survey reports are submitted according to the Coral PDCs (Appendix C).

5) Requiring relocation of ESA-listed corals. The tally of these projects will include the total number and type of ESA-listed corals relocated by species and a summary of the survival rates for the year, according to the Coral PDCs (Appendix C).

6) Project activities located within the range of Johnson's seagrass that required a survey. The tally of these projects will include a summary of the results of the post-construction surveys.

2.9.4.3 Lessons Learned

The annual programmatic review and report will include feedback on the unique situations encountered for projects covered under this Opinion and how they were resolved. These lessons learned are valuable to understanding if minimization measures worked, predicating the likelihood of encounters or take of ESA-listed species in specific areas depending on variables such as time of year or site conditions, and identifying where clarification or training may be necessary for issues related to this Opinion. The format in which these comments provided is at the discretion of the USACE in coordination with BOEM and can be as simple as adding comments fields to a spreadsheet tracking all projects covered under this Opinion. Some of this information may be gathered while the project is active or comments about information discovered while reviewing all projects completed and compiled for the programmatic annual review. Important information includes:

1) Corrective action taken during construction of a project.

[20] Note that adverse effects to designated critical habitat are not anticipated as a result of the proposed action; however, this reporting requirement ensures that NMFS will be notified in the event that adverse effects to critical habitat have occurred.

2) Information gathered during the risk-based adaptive management process including species trends and use of an area; especially if it resulted in more or less take than expected at a specific project location.
3) Lessons learned based on site-specific conditions observed during a project that may be relevant to future projects (e.g., difficulty keeping the hopper dredge dragarm firmly embedded due to site conditions).
4) A summary of successes and challenges encountered during projects conducted under the alternative review process (Section 2.9.5 of this Opinion).
5) Discrepancies observed between USACE Districts on the interpretation of PDCs to determine if a project should be covered under SARBO and the corrective action taken to resolve the inconsistency.

2.9.5 Alternative Project Implementation and Programmatic Modification through the Superseding Process of Review

While NMFS, USACE, and BOEM have worked to collaboratively develop the PDCs in this Opinion, instances may arise where a project, methodology, or equipment type does not exactly fit in the scope or scale of work defined by the PDCs, where the PDCs require modification to operate as intended, or where project-specific review by NMFS is required under the PDCs.

This Opinion considers limited flexibility in the implementation of projects covered under this Opinion, as well as for revisions to the PDCs. In these instances, the USACE and/or BOEM may propose to use new materials or equipment methods not considered in this Opinion or propose a project that may deviate from the PDCs in a minor fashion. The USACE or BOEM must first determine that the modification will have effects on ESA-listed species or designated critical habitat that are "substantially similar" to the effects considered in this Opinion, and submit its determination to NMFS. NMFS will consider effects to be "substantially similar" if the effects of the proposed project are consistent with the effects analyzed in this Opinion (i.e. does not result in effects to an ESA-listed species or critical habitat not considered in the Opinion). NMFS will not approve any project under this alternative review process that the agency determines may have effects greater in magnitude or scale than those analyzed in this Opinion, or where reinitiation is warranted. Elements of PDCs which minimize the impacts of take of ESA-listed species may not be modified to diminish the minimization of the impacts of such take. This alternative review process is intended to allow limited modifications, while ensuring that effects remain consistent with those analyzed in this Opinion.

Through the processes described in this section, USACE and/or BOEM may request NMFS review and approval of:

1. Individual projects that deviate slightly from the PDCs (provided in Appendix B - Appendix H) (as outlined in Section 2.9.5.1 of this Opinion – *SARBO Supersede 1. Superseding Process for Review and Inclusion of Substantially Similar Projects or Projects with Substantially Similar Effects*), or require project-specific review under the PDCs.
2. New methods of conducting activities or equipment types that deviate from the proposed action in this Opinion (as outlined in Section 2.9.5.2 of this Opinion –*SARBO Supersede 2.*

Superseding Process for Review and Inclusion of New Construction Methodologies or Equipment Types with Substantially Similar Effects).

3. Minor modifications to specific existing PDCs for future projects covered under this Opinion (as outlined in the Section 2.9.5.3 of this Opinion - *Adaptive Changes to PDCs*).
4. Project-specific review for relocation of ESA-listed corals

2.9.5.1 SARBO Supersede 1. Superseding Process for Review and Inclusion of Projects with Substantially Similar Effects

If the USACE or BOEM makes the preliminary determination that a proposed project deviates from the PDCs, but the effects would be substantially similar to the effects considered in this Opinion, it may provide the rationale to NMFS and request permission to rely on the Opinion to satisfy its ESA Section 7 consultation obligations. The request to use the procedures in this Section must be accompanied by supporting documentation explaining the project, including:

1) Project description and rationale why the effects of the proposed action are substantially similar to the analysis in this Opinion, as shown in the examples in Table 7.
2) Relevant construction drawings, benthic surveys, survey data, or other information supporting the determination.
3) For use of an equipment type or construction methodology not considered in this Opinion, for a specific project, documentation (e.g., reports or studies) supporting the conclusion that the effects from the proposed modification would be consistent with or less than those evaluated in this Opinion.

When requesting consideration of a project under the alternative review process, the USACE and BOEM will email the request to SERO.dredge@noaa.gov and must await written approval from NMFS (email) before authorizing the project. These projects will be listed in the annual report including documentation from the USACE and/or BOEM on the successes and lessons learned from the completion of these projects. For consistency and to better track the types of requests for an alternative review process, requests from USACE District Offices will be coordinated through the USACE SAD according to their procedures, unless modified overtime by the USACE SAD.

NMFS will provide its determination that that the effects of a specific proposed project under modified PDCs are or are not substantially similar to the effects discussed and found in this Opinion, to the lead action agency (e.g., USACE or BOEM) for that specific project.

Table 7. Example of Requests that may be submitted under SARBO Supersede and the rationale for potential approval

Date Sent	Action Agency Tracking Number	Project Name	Latitude	Longitude	Critical Habitat (list unit or N/A)	PDC that modification is requested	Brief project description and rationale why the Supersede is requested.	Explanation why the modification is still substantially similar to the effects considered in this Opinion
01/01/2020	N/A	ABC Channel	#.####	-#.####	N/A	PDC HOPPER.5	The PDC requires that a state-of-the-art solid-faced deflector is attached to the draghead during hopper dredging. The dredge area has large debris present and use of the deflector shield would be hazardous and ineffective.	The deflector shield will only need to be removed for the beginning of the project (anticipated to be complete in less than 2 weeks) and will be reattached once the large debris is removed. Dredging will occur when the risk of take of [list species] is low due to the time of year [insert time of year work will occur and rationale why the decreases the risk of take]. This area has been dredged without a deflector shield during similar past events (i.e., [list history]) that did not result in an increase in take. Based on these factors, USACE believes that removal of dragheads at the beginning of the project will not increase the risk of take of ESA-listed species. If take occurs, dredging will stop and NMFS will be contacted for a determination if the "substantially similar effects" determination is still appropriate.
01/01/2020	SAS-2018-#####	ABC Beach	#.####	-#.####	N/A	PDC C-BEACH.1	Beach nourishment is limited to placement within the fill template	Beach nourishment is proposed at a beach within the range of ESA-listed corals that has **not** been previously nourished, but was severely eroded by a recent hurricane [insert hurricane name and year]. Google Earth

Date Sent	Action Agency Tracking Number	Project Name	Latitude	Longitude	Critical Habitat (list unit or N/A)	PDC that modification is requested	Brief project description and rationale why the Supersede is requested.	Explanation why the modification is still substantially similar to the effects considered in this Opinion
							provided in SARBA Appendix B or a previous Section 7 consultation.	shows the limits of the beach for the last 10 years and the applicant wishes to fill to the previous limit (ETOF). No seagrass or hardbottom is present in the area, it is not in designated critical habitat. It is a sea turtle nesting beach, but placement of sand will occur outside of nesting season.. All other PDCs will be followed. Based on these factors, USACE believes that placement of fill within the previous ETOF will not result in effects beyond those considered in SARBO.
	SAC-2019-#####	River 2 dredging	#.####	-#.####	N/A	PDC Sturgeon.3	Dredging in the river is proposed a half mile upriver of the upper river work limit identified in Table 56 in Appendix E. The applicant proposes to complete dredging in this stretch of river in approximately 2	Dredging is proposed in an area of the river above the upper river limits in this Opinion. The USACE believes that dredging in this area at the time proposed will not result in effects to shortnose or Atlantic sturgeon beyond those considered in 2020 SARBO. Work will occur in November when water temperature and dissolved oxygen levels are expected to be within the range established for sturgeon by the SARBO PDCs, based on [cite source of water quality information], and the project is located in a side channel off a main river where sturgeon are not expected to occur. Atlantic sturgeon critical habitat is not

Date Sent	Action Agency Tracking Number	Project Name	Latitude	Longitude	Critical Habitat (list unit or N/A)	PDC that modification is requested	Brief project description and rationale why the Supersede is requested.	Explanation why the modification is still substantially similar to the effects considered in this Opinion
							weeks by mechanical dredging with upland placement.	designated in or near the proposed dredging area and therefore will not be effected. USACE accordingly believes that dredging in this location during the proposed time will not result in any additional effects to ESA resources beyond those in SARBO.

2.9.5.2 SARBO Supersede 2. Superseding Process for Review and Inclusion of New Construction Methodologies or Equipment Types with Substantially Similar Effects

The USACE and BOEM propose to continue to coordinate with and engage the dredging industry in the development of innovative solutions to improve dredging efficiencies and to reduce the risk of incidental take. As part of this effort, USACE/BOEM may request to incorporate new construction methodologies or new equipment types (including equipment modifications) into future dredging projects covered under this Opinion. NMFS will review these requests to determine if the effects expected are substantially similar to those considered in this Opinion.

Any request to use the SARBO Supersede 2 review option would require sufficient documentation and rationale to support a "substantially similar" determination when considering the effect to ESA-listed species and designated critical habitat. Relevant information to evaluate a new construction methodology or an equipment type should include documentation such as the successful use of it in other projects, studies completed on the effectiveness of it, and documentation from industry experts on the merits and concerns of the new methodology or equipment type. Innovative designs may also be considered if sufficient rationale can be provided to demonstrate that the effects would still be substantially similar to those considered in this Opinion. USACE and BOEM will also need to provide NMFS with the contingency plan if incidental take were to occur with the new equipment type.

When requesting consideration of a project under the Supersede 2 process, the USACE and BOEM must await written approval from NMFS before authorizing the project. All projects completed using the new construction methodologies or equipment types approved by NMFS using the SARBO Supersede 2 process will also be listed in the annual report including documentation from the USACE and/or BOEM on the successes and lessons learned from the completion of these projects. If new information becomes available indicating that the effects of these modifications are not "substantially similar" to those types evaluated in this Opinion, NMFS will alert the USACE and/or BOEM that such construction methodologies or equipment types can no longer be used or must be reconsidered before further use.

Examples of scenarios in which the SARBO Supersede 2 review process may be used under this Opinion and basis for determining that the project's effects would be substantially similar to those analyzed in this Opinion:

1. <u>Hopper dredge deflector shields</u>: The General PDCs in Section 3.1 in Appendix B states that a state-of-the-art solid-faced deflector that is attached to the draghead must be used on all hopper dredges at all times. However, exceptions could be considered if the USACE and/or BOEM demonstrate that the effects to ESA-listed species from a new design are consistent with the effects analyzed in this Opinion.
2. <u>New Avoidance Technology</u>: The USACE is testing the use of a new technology that disturbs the area in front of a hopper dredge draghead to move ESA-listed species in its path out of harm's way. If testing and studies sufficiently demonstrate this equipment modification reduces the risk of take and all other effects from the use of this new equipment are consistent

with those analyzed in this Opinion, then the addition of the equipment type may be added as an option for future projects covered under this Opinion.

New technologies considered under Supersede 2 may be reported in the Supersede 1 sample reporting table (Table 7) for consideration of use on a specific project. If the new technology is successful and frequently requested to be used on projects covered under this Opinion, NMFS will consider using the process outlined next in Section 2.9.5.3 of this Opinion to add it to the list of approved equipment types covered under this Opinion for all projects.

2.9.5.3 Adaptive Changes to PDCs

The SARBO Team may determine that a minor revision to 1 or more PDCs may be necessary, particularly where (1) the PDC(s) is/are not operating or being interpreted in the manner intended, (2) updated information is available to improve guidance and/or specifications in the PDCs, or (3) new technologies are available that allow for a reduction of impacts of takes to ESA-listed species, including the likelihood of take.

PDCs for this proposed action include elements that minimize the impact of takes of ESA-listed species. A PDC will only be changed if there is sufficient information to support the proposed change, as determined by NOAA Office of General Counsel Southeast Region and the NMFS Southeast Regional Administrator, and only if the Regional Administrator determines that the change would not diminish the minimization of impacts of takes to ESA-listed species, nor trigger reinitiation of this Opinion under 50 CFR Section 402.16. In the event that a change to a PDC is approved, a revised version of the Appendix with the revised PDC will be uploaded to both ODESS and the NMFS Southeast Region website (**SERO.Dredge@noaa.gov**) and all future projects covered under this Opinion will be required to adhere to the new PDCs.

The following examples illustrate new information obtained that would not be expected to trigger reinitiation but may result in a PDC revision:

Revisions to survey requirements: Both the coral and Johnson's seagrass PDCs include protective measures intended to minimize and avoid adverse effects from turbidity and sedimentation to these non-mobile ESA-listed species (i.e., these species cannot avoid areas with increased turbidity). The survey distances stated in the PDCs are based on the best available information at the time that the Opinion was completed. Projects completed under this Opinion using the defined surveying and reporting requirements will be evaluated periodically to better understand the range of turbidity and sedimentation associated with projects covered under this Opinion. If multiple years of monitoring demonstrates that turbidity and sedimentation either for all projects or in a specific location do not extend beyond the dredge or material placement footprint, or that the extent of the effects is considerably less than the required survey distances, USACE may request that NMFS reevaluate the survey distances necessary to ensure that sedimentation and turbidity effects are adequately monitored. The Johnson's Seagrass PDCs include various dredging methods and turbidity curtain requirements. If the turbidity and sedimentation are adequately maintained by the methods outlined in the PDCs, the requirement to conduct pre- and post-construction surveys may be removed using this adaptive changes to PDC process. In both of these examples, USACE would first notify NMFS of its determination that modifying surveying/reporting requirements would not be expected to change effects to

ESA-listed species or critical habitat. Before the PDCs are modified or removed, a review would be conducted of the survey results from the projects completed in the area and any other available information to ensure that the effects to ESA-listed corals, Johnson's seagrass, or coral hardbottom from turbidity and sedimentation associated with dredging and/or material placement are still consistent with the effects considered in this Opinion.

<u>Revision to water quality protection measures</u>: Water quality protections in sturgeon rivers (Appendix E) are designed to be protective to Atlantic and shortnose sturgeon during times based on historical records of when and where water temperatures are high, dissolved oxygen (DO) is low, and additional stress to sturgeon from water quality changes from dredging could be harmful. These protective measures are based on the best available information with acknowledgement that information on dredge impact to DO in sturgeon rivers is limited and water quality in these rivers may change over time. The USACE and NMFS agree to continue to review studies completed and available water quality information in these rivers and adjust the seasonal windows defined in the Sturgeon PDCs in Appendix E as necessary, to ensure that the water quality effects from future projects are consistent with the analysis in this Opinion.

<u>Revision to the upper river limit of dredging covered under this Opinion and the location of aggregation areas in sturgeon rivers</u>: The Sturgeon PDCs establish an upper river work limit in these rivers to be protective of potential spawning habitat and identifies aggregation areas used by sturgeon in rivers so that work does not occur in sensitive areas. These limits are based on the best available information on how and where sturgeon use these rivers and is intended to be updated as new information becomes available. Once the presence and absence of sturgeon spawning areas are better known in particular areas, USACE may request that the upper river limits for dredging, which are designed to protect spawning habitat, be modified in accordance with the updated information. Because the Opinion currently limits any activities based on the potential presence of spawning habitat and aggregations, this change would not be expected to change any of the analysis in this Opinion as the protective provisions for sturgeon spawning habitat and aggregations would apply to any identified spawning areas.

2.9.5.4 Project-Specific Review for Relocation of ESA-listed Corals

Under the PDCs, limited circumstances may arise where relocation of ESA-listed corals is appropriate to minimize the risk of lethal take from beach nourishment or pipeline placement. If USACE identifies ESA-listed corals within (1) 500 ft of the ETOF for beach nourishment projects, or (2) the 25 ft pipeline corridor for pipeline projects, USACE will contact NMFS to conduct a project-specific review of anticipated impacts to ESA-listed corals and to determine if relocation is appropriate under this Opinion.

The project-specific review request will be accompanied by supporting documentation explaining the project, including:

1) Relevant construction drawings, benthic surveys, and survey data (including surveys completed consistent with 2020 SARBO Appendix C)
2) Explanation of anticipated impacts to ESA-listed corals and coral hardbottom

When requesting consideration of a project under the project-specific review process, the USACE will email the request to **SERO.dredge@noaa.gov** and await written approval from NMFS (email) before authorizing the project. NMFS will provide its determination of whether relocation is an appropriate minimization measure and consistent with the effects considered in this Opinion to the lead action agency (e.g., USACE or BOEM) for that specific project. These projects will be listed in the annual report including documentation from the USACE and/or BOEM on the successes and lessons learned from the completion of these projects. For consistency and to better track the types of requests for an alternative review process, requests from USACE District Offices will be coordinated through the USACE SAD according to their procedures, unless modified overtime by the USACE SAD.

3 Potential Routes of Effect to ESA-listed Species and Critical Habitat

Section 3 identifies each route of effect that we believe will occur as a result of the activities covered under this Opinion, as described and limited by the PDCs in Appendix B-Appendix H. Section 3.1 of this Opinion discusses the potential routes of effects to mobile ESA-listed species; Section 3.2 of this Opinion discusses potential routes of effects to non-mobile ESA-listed species, and Section 3.3 of this Opinion discusses the potential routes of effects to designated critical habitat. Each route of effect is described in Section 3 and includes our determination of whether the effect to each identified ESA-listed species and designated critical habitat is: no effect; may affect, not likely to adversely affect; or may affect, likely to adversely affect. No effect (NE) or may affect, not likely to be adversely affect (NLAA) determinations are not analyzed beyond Section 3 of this Opinion. We believe that none of the activities covered under this Opinion will result in adverse effects to designated critical habitat and therefore designated critical habitat is not discussed beyond Section 3 of this Opinion. Effects that we believe may affect and are likely to adversely affect (LAA) ESA-listed species are analyzed further in Section 6.

Table 8 and Table 9 provide the USACE and BOEM's final effects determinations for ESA-listed species and designated critical habitat, respectively, based on their evaluation of the activities occurring within the action area. The USACE and BOEM initially provided different determinations with their request for consultation, but revised those determinations after completing the PDCs and re-evaluating the effects to ESA-listed species. The USACE and BOEM divided their effects determinations for each species into 3 categories (hopper dredging effects, relocation trawling effects, and effects from all other activities) for all ESA-listed species, except corals and whales, as shown in Table 8.

Table 8 and Table 9 also provide NMFS' final effects determination for each ESA-listed species and designated critical habitat. NMFS effects determinations in Table 8 and Table 9 reflect our overall determination for all effects analyzed in the Opinion. For example, if we believe that an activity may have no effect on a species and another activity may affect that species, but is not likely to adversely affect it, the NMFS final effects determination is that the proposed action may affect, but is not likely to adversely affect that species.

Table 8. Effects Determination(s) for Species the Action Agencies and/or NMFS Believe May Be Affected by the Proposed Action

ESA-listed Species	ESA Listing Status[21]	USACE/BOEM[22] Hopper Dredging (HD) Relocation Trawling (RT) All Other (OT)	NMFS (Final)
Sea Turtles			
Green (North Atlantic [NA] DPS)	T	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Green (South Atlantic [SA] DPS)	T	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Hawksbill	E	LAA$^{HD, RT}$, NLAA^{OT}	NLAA
Kemp's ridley	E	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Leatherback	E	LAA^{RT}, NLAA$^{HD, OT}$	LAA
Loggerhead (Northwest Atlantic [NWA] DPS)	T	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Fish			
Atlantic sturgeon (Carolina DPS)	E	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Atlantic sturgeon (SA DPS)	E	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Atlantic sturgeon (Gulf of Maine DPS)	T	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Atlantic sturgeon (New York Bight DPS)	E	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Atlantic sturgeon (Chesapeake Bay DPS)	E	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Shortnose sturgeon	E	LAA$^{HD, RT}$, NLAA^{OT}	LAA
Nassau grouper	T	NLAA$^{OT,HD, RT}$	NLAA
Elasmobranchs			
Giant manta ray	T	LAA^{RT}, NLAA$^{HD, OT}$	LAA
Scalloped hammerhead shark (Central and Southwest Atlantic DPS)	T	LAA^{RT}, NLAA$^{HD, OT}$	NLAA
Smalltooth sawfish (U.S. DPS)	E	LAA^{RT}, NLAA$^{HD, OT}$	LAA
Oceanic whitetip shark	T	NE	NE
Whales			
Blue whale	E	NE	NLAA
Fin whale	E	NE	NLAA
North Atlantic right whale	E	NLAA	NLAA
Sei whale	E	NE	NLAA
Sperm whale	E	NE	NLAA
Non-Mobile Species (corals and marine plants)			
Boulder star coral (*Orbicella franksi)*	T	LAA	LAA
Elkhorn coral (*Acropora palmata*)	T	LAA	LAA
Lobed star coral (*Orbicella annularis*)	T	LAA	LAA
Mountainous star coral (*Orbicella faveolata)*	T	LAA	LAA
Pillar coral (*Dendrogyra cylindrus*)	T	LAA	NLAA
Rough cactus coral (*Mycetophyllia ferox*)	T	LAA	NLAA
Staghorn coral (*Acropora cervicornis*)	T	LAA	LAA
Johnson's seagrass	T	LAA	LAA

[21] E= endangered; T= threatened; NE = no effect

[22] USACE/BOEM used MALAA (may affect, likely to adversely affect) and MANLAA (may affect, not likely to adversely affect), shortened to LAA and NLAA to match NMFS acronyms.

Oceanic whitetip shark
We believe there will be no effect to oceanic whitetip shark from activities covered under this Opinion. This pelagic species is found throughout the world in tropical and sub-tropical waters, generally remaining offshore in the open ocean, on the outer continental shelf, or around oceanic islands in water depths greater than 600 ft. They live from the surface of the water to at least 498 ft deep. Oceanic whitetip sharks have a strong preference for the surface mixed layer in warm waters above 20°C, and are therefore a surface-dwelling shark.[23] While some activities covered under this Opinion will occur in offshore federal waters in ODMDS and borrow sites, these locations are generally in waters less than 100 ft deep, which is shallower than the known range of this species. Activities covered under this Opinion also include projects in federal waters off the U.S. Virgin Islands and Puerto Rico, which are oceanic islands listed as a potential habitat location for this species; however, work occurring in these areas are still expected to occur in waters shallower than those occupied by this species. While we believe that they will not occur within the action area, handling measures for all elasmobranchs (including shark species such as the Oceanic whitetip) are included in this Opinion with the expectation that Oceanic whitetip shark will not be encountered. The effects analysis in this Opinion will not consider this species further.

Scalloped hammerhead shark – updated information
At the time we listed the Central and Southwest Atlantic DPS of scalloped hammerhead shark, it was thought to occur within the U.S. Caribbean (79 FR 38213, Publication Date July 3, 2014). However, in designating critical habitat for the Central and Southwest Atlantic DPS of scalloped hammerhead shark the following year (80 FR 71774, Publication Date November 17, 2015), we determined that there is no evidence that the scalloped hammerhead shark was present within the waters of the U.S. Caribbean. Since that time, we have received additional information indicating that this species does occur in waters of the U.S. Caribbean. In connection with our consultation on fishery independent monitoring in the U.S. Caribbean, we learned of a single recorded interaction during hook-and-line fishery monitoring (NMFS 2016a), and are now aware of a limited number of additional interactions. For purposes of this Opinion, we believe that the Central and Southwest Atlantic DPS of scalloped hammerhead shark is found in waters of the U.S. Caribbean, among other areas, but is not found in waters off Florida. Therefore, we consider effects to this DPS only in waters off Puerto Rico and the U.S. Virgin Islands and do not consult on effects to this DPS in waters off Florida.

NMFS identified a population of scalloped hammerhead shark that occurs in waters off Florida (i.e., the Northwest Atlantic and Gulf of Mexico DPS for scalloped hammerhead shark); however, we determined that DPS did not warrant listing (78 FR 20717, Publication Date April 5, 2013 and 79 FR 38213, Publication Date July 3, 2014).

[23] https://www.fisheries.noaa.gov/species/oceanic-whitetip-shark

Table 9. Effects Determinations for Designated Critical Habitat the Action Agency and/or NMFS Believe May Be Affected by the Proposed Action

Species	Critical Habitat	USACE/ BOEM[24]	NMFS (Final)
Green sea turtle (NA and SA DPS)	Culebra Island	NE	NE
Leatherback sea turtle	Sandy Point, St Croix and U.S. Virgin Islands	NE	NLAA
Loggerhead sea turtle (NWA DPS)	Nearshore Reproductive Habitat (Units LOGG-N-3 to LOGG-N-20), Breeding Habitat (Units LOGG-N-17 and LOGG-N-19), Migratory Habitat (Units LOGG-N-17 to LOGG-N-19), *Sargassum* Habitat (Units LOGG-S-01), Winter Habitat (Units LOGG-N-1 and LOGG-N-2)	NLAA, NE for *Sargassum* Habitat	NLAA
Hawksbill sea turtle	Mona and Monita Island	NE	NE
Atlantic sturgeon	Carolina Units 1-7 and South Atlantic Unit 1-7	NLAA	NLAA
Acropora (Elkhorn and staghorn coral)	Florida and Caribbean Units	NLAA	NLAA
Johnson's seagrass	Units A-J	NLAA	NLAA
North Atlantic right whale	Unit 2	NE	NE

[24] NE = no effect; NLAA = may affect, not likely to adversely affect; LAA = may affect, likely to adversely affect. USACE/BOEM NLAM (may affect, not likely to adversely modify) determinations shortened to NLAA to match NMFS acronyms.

3.1 Potential Routes of Effects to Mobile ESA-listed Species

This Section analyzes the effects of the proposed action to mobile ESA-listed species that may occur within the action area (sea turtles, fish, elasmobranches, and whales listed in Table 8). Section 3.2 of this Opinion analyzes those ESA-listed species that are non-mobile (Johnson's seagrass and ESA-listed coral and listed in Table 8**).**

In Section 2 of this Opinion, the activities analyzed under this Opinion were divided into 5 general categories (listed below).

1. Dredging (including all forms of dredging discussed in Section 2.3 of this Opinion) and related activities, such as relocation trawling. For the effects analysis below, geotechnical surveys (discussed in Section 2.6) will be considered to have the similar effects as mechanical dredging since this activity removes material by taking samples of sediment, though the effects of a onetime sediment sample are smaller in scope and scale.
2. Placement of dredged materials (including all forms discussed in Section 2.4 of this Opinion)
3. Vessels used for dredging, transportation, and material placement (discussed in Section 2.5)
4. Geophysical surveys authorized by the USACE necessary to complete dredging and material placement projects (discussed in Section 2.6 of this Opinion).
5. Monitoring for and handling of ESA-listed species encountered during projects covered under this Opinion (discussed in Section 2.7 of this Opinion).

In this Section, we also consider how each of the 5 categories of activities listed above may affect ESA-listed species by analyzing the potential routes of effects expected to occur from those 5 categories of activities by considering:

- Species interactions with dredging and material placement equipment, including entrainment or impingement[25] and the potential for effects from degraded water quality (Section 3.1.1).
- Potential entanglement with equipment (Section 3.1.2).
- Impacts caused by capture via relocation and abundance trawling (Section 3.1.3).
- Potential for a species to be struck by a vessel (Section 3.1.4).
- How species interact with the placement of material (Section 3.1.5 of this Opinion).
- The potential for blocked access by construction activities (Section 3.1.6 of this Opinion).
- Habitat alteration for activities covered under this Opinion (Section 3.1.7 of this Opinion).
- Sound generated by activates covered under this Opinion (Section 3.1.8 of this Opinion).

[25] For this Opinion, entrainment occurs when a species either comes into contact with a suction type dredge (hopper or cutterhead) or is in close enough proximity that they cannot outswim the suction velocity created by the dredge. Impingement occurs when the species is captured by the equipment (e.g., captured in a mechanical dredge) or stuck to the equipment (e.g., entrained by a hopper dredge, but stopped by grating on the draghead that prevents movement into the hopper).

3.1.1 Species Interaction with Dredging and Material Placement Equipment Covered under this Opinion

This Section evaluates the potential for injury to mobile ESA-listed species that may be present in the action area from physical interactions with equipment used for dredging and material placement under this Opinion (e.g., mechanical, cutterhead, hopper, and agitation dredging equipment), or due to potential effects caused by changes in water quality due to dredging.

3.1.1.1 Data Supporting Determinations in this Section Regarding the Potential for Avoidance of Physical Impacts

While we have previously identified and confirmed routes of adverse effect from hopper dredging (as discussed below in Section 3.1.1.5 and further in Section 6), we have historically assumed that mobile species will avoid many potential routes of effect associated with non-hopper dredging, based on their ability to avoid activities if the activity is slow moving (e.g., mechanical and cutterhead dredging, which is generally barge-mounted, works in a set location, and then stops working while being relocated from 1 dredging location to another). In previous Opinions, we have explained that we believed that it was extremely unlikely that a mobile species would remain in an area where non-hopper dredging activities were taking place, and therefore would avoid interactions with these activities, other than the potential for vessel strikes. Otherwise, we considered the likelihood of an interaction between non-hopper dredging activities and a mobile species to be so low that we considered the route of effect to be discountable, since mobile species are likely to avoid construction noise, moving equipment, and placement or removal of materials during construction, making interactions extremely unlikely.

Given the programmatic nature of this Opinion, and the large scale of dredging expected to occur as part of the activities considered by this Opinion, we decided to evaluate our assumptions regarding avoidance behavior, based on available research papers and reported interactions between non-hopper dredging equipment and ESA-listed species. We have no reported take caused by non-hopper dredging equipment for most mobile ESA-listed species, including elasmobranchs (giant manta ray, scalloped hammerhead, and smalltooth sawfish), Nassau grouper, and whales (blue, fin, sei, sperm, and North Atlantic right). However, we discovered some reported takes of ESA-listed sturgeon and sea turtles associated with non-hopper dredging projects, which are discussed by activity type below.

Studies also indicate that Atlantic and shortnose sturgeon may be unlikely to avoid dredging activities, and may respond to the presence of dredging differently in riverine environments than in open ocean areas, as discussed below.

Reine et al. (2014) tagged 5 Atlantic sturgeon to monitor their movements in the presence of a cutterhead dredge in a river. Two animals ultimately vacated the areas around the dredge traveling 12-28 miles from the operation, though the authors noted one of the animals showed no avoidance behavior when near the dredge (Reine et al. 2014). Two other animals remained in the area where dredging was occurring, moving back and forth across the river segment being dredged showing no apparent changes in behavior that could be assigned to the dredge. The last animal spent 5 days near the area being dredged, at one point passing within 50 m of the dredge itself, before leaving and heading several miles upstream (Reine et al. 2014). While it must be acknowledged the sample size of this study is small, the authors conclude there was no evidence that an active dredging operation represented a physical barrier to sturgeon movement (Reine et al. 2014). They also report Atlantic sturgeon did not show either attraction or avoidance responses to the physical presence of the dredge itself, noise generated during the dredging operation, or disturbance of sediment, either from increase turbidity or resuspending potential food resources in the water column (Reine et al. 2014). DiJohnson (2019) came to a similar conclusion after monitoring Atlantic sturgeon in the Delaware River. The author concluded they showed no behavioral response to vessel traffic and even suggested they may lack the ability to avoid vessel traffic (DiJohnson 2019).

Parsley et al. (2011) provide further evidence that sturgeon appear unaffected by increased turbidity associated with hopper dredging. The authors report a sturgeon species closely-related to Atlantic and shortnose (white sturgeon) was exposed to increased turbidity from hopper dredging, but that exposure did not prompt an avoidance behavior (Parsley et al. 2011). The authors postulate the change in underwater noise and increase suspended material may have actually spurred those animals to investigate the area (Parsley et al. 2011). They even report a single individual held its position within a disposal site during and after a dredge was actively offloading dredge spoils before leaving (Parsley et al. 2011). In essence, the animal held its position in the disposal site as dredge material rained down on it. It is unclear how applicable this observed behavior is to Atlantic and shortnose sturgeon, but this paper provides clear evidence that a closely-related species did not relocate from, and potentially sought out, areas of high turbidity resulting from dredging. Therefore, we evaluate the potential effects to Atlantic and shortnose sturgeon from dredging and material placement in the effects analysis for all activities covered under this Opinion and analyzed in Section 3 of this Opinion, based on the studies described above as well as a review of reported interactions by equipment type.

3.1.1.2 Data Supporting Determinations in this Section Regarding the Potential for Changes in Water Quality to Effect Mobile Species

This section discusses the information and data used to analyze the potential routes of effect to mobile ESA-listed species from water quality changes resulting from dredging and material placement covered under this Opinion, with a particular focus on available data for Atlantic and shortnose sturgeon in rivers within the action area, due to that species' particular sensitivity to changes in water quality in riverine environments where spawning occurs.

Dredging and material placement covered under this Opinion may affect species and habitat in and around the project footprint through changes in water quality. The suspension of sediment in the water column during dredging and material placement can result in turbidity in the area that may impact animals and reduce water clarity needed for photosynthesis of plants.

The type of dredging equipment used can result in varying levels of turbidity, total suspended solids, and sedimentation. Dredging equipment is generally designed to scoop (e.g., mechanical dredges such as clamshell and bucket dredges), suction up (e.g., cutterhead pipeline and hopper dredges), agitate the sediment to resuspend solids (e.g. water injection dredging), or to smooth over/level out sediments (e.g., bed-leveling). The disposal method of dredged sediments can also affect turbidity such as side cast that sprays sediments to the side of the dredging, hopper dredge overflow that allows water to run off of the sediment collected in the hopper, or beach nourishment projects where sand is placed on the beach or nearshore environment to feed the beach.

This Opinion relies on scientific literature, and information provided by the NOAA Greater Atlantic Region regarding the expected effects for turbidity and total suspended solids (https://www.greateratlantic.fisheries.noaa.gov/protected/section7/guidance/consultation/turbidit ytablenew.html), including information on newer technologies such as water injection dredging and bed-leveling, for distances that suspended solids may extend from a dredging project are based. NOAA Greater Atlantic Region's turbidity analysis is appropriate to consider for the action area because the mechanisms by which turbidity is created (i.e., dredging), the physics of turbidity (i.e., how it moves through water), and its routes of effect to species (e.g., potential abrasion) are the same across the regions.

Based on the information reviewed, generally dredging and material placement-generated turbidity plumes are limited to an area only a few hundred feet to a few thousand feet and most turbidity settles out quickly once dredging or material placement is complete. However, fine grain sediments can travel further distances and stay suspended longer than heavier material like sand. For example, the potential for large turbidity plumes was demonstrated by dredging in the Port of Miami deepening project, which penetrated coral hardbottom and generated fines that were directly observed at least 2.5 kilometers (km) from the channel, with predicted impacts (based on sediment plume occurrence) as far as 10–15 km away (Cunning et al. 2019). We do not expect turbidity and sedimentation effects to this scale from dredging or material placement covered under this Opinion based on the restrictions contained in the PDCs.

Suspended solids (either measured as turbidity or total suspended solids) can be carried to adjacent locations and result in sedimentation that can cover or bury nearby non-mobile species and habitat such as seagrasses and corals. The distance suspended solids can travel outside of the project footprint can vary dramatically depending on the density of the suspended solids (generally referred to as the percent of fines in the material) and local hydrographic patterns, such as the local tides and currents. The velocity of water movement in the area can affect the time that suspended solids remain in the area. For example, riverine environments with an outgoing tide will flush away turbidity quicker than areas with less current such as an estuary with limited tidal flushing. In rivers, the currents also act to compress the turbidity plume as it moves downstream and settles, reducing the overall area/volume affected by it (see Figure 15 for an example). Higher velocity currents may spread suspended solids to a larger area than if they were to settle out closer to the dredge or material placement footprint. Burton (1993) measured elevated total suspended solid concentrations up to 3,000 ft away from dredge sites in the Delaware River. We anticipate elevated total suspended solid concentrations could travel this distance in rivers within the action area. The suspension of solids in the water can also cause other water quality concerns such as changes in the amount of available DO and temperature.

Changes in water quality conditions (e.g., water temperature or DO concentrations) can affect the physiological capacity of mobile species to respond to dredging and dredging-related impacts. This is especially true for sturgeon in rivers. Therefore, we consider not only the effects to these species from different types of dredging, but also how the water quality conditions at the time of dredging, and the changes to those conditions caused by dredging, effect these species.

Atlantic and shortnose sturgeon are affected by the water quality of the rivers in which they live and spawn (referred to in this Opinion as “sturgeon rivers” listed in Table 56 of Appendix E). We believe the water quality conditions in sturgeon rivers affect how they respond to dredging and the dredging-induced changes in water quality. Information gathered to complete this Opinion included, among other things, a review of the historical water temperature and DO concentrations measured in these rivers. We also considered telemetry data to identify areas of high occupancy and likely seasonal aggregations. In gathering information on water quality in sturgeon rivers, it became apparent that the majority of sturgeon rivers in the Southeast suffer from naturally occurring high water temperatures (e.g., 28-30°C) and low DO concentrations (e.g., less than 4.3 mg/L) during the summer months.

Some rivers in the action area (e.g., Savannah River, Cooper River, Neuse River) consistently have very low DO concentrations in the summer, to the point that portions of the river become hypoxic or anoxic. Hypoxic zones as areas in aquatic systems of such low oxygen concentration that animal life suffocates and dies, and as a result are sometimes called “dead zones.” Hypoxia, or low DO concentrations, occurs when the amount of DO in water becomes too low to support most aquatic life (typically below 2 mg/L). Hypoxia can occur naturally but it is often a symptom of degraded water quality resulting from man-made activities (e.g., nutrient pollution). Common sources of nutrient pollution include agricultural runoff, the burning of fossil fuels, and wastewater discharges. Over half of U.S. estuaries experience natural or human-induced hypoxic conditions at some time each year and the frequency and duration of hypoxic events have increased exponentially over the last few decades. Studies conducted in the Savannah River in July and August 2019, by the USACE ERDC to determine changes in DO from cutterhead dredging measured DO as low as 0.7 mg/L DO. The report notes there is a considerable low DO zone from the Route 17 Bridge to the Houlihan Bridge, which may shift based on tide. Low DO similar to these were measured by South Carolina Department of Natural Resources in May of 2019 (USACE 2019b).

Other rivers in the action area have also been observed to be anoxic after stochastic events such as Hurricane Florence that resulted in flooding that flushed significant amounts of organic matter into rivers supporting sturgeon in the Southeast. The DO levels in those rivers dropped so low (i.e., 0.2 mg/L) that thousands of fish suffocated, including multiple sturgeon.

Periods of low DO concentrations and high water temperature, can result in physiological stress (Campbell and Goodman 2004; Jenkins Jr. et al. 1993; Secor and Gunderson 1998; Secor et al. 2000) and poor body condition (Flournoy et al. 1992b) for sturgeon. Stress symptoms may include immobility or reduced movement (Jenkins et al. 1993)(Crocker and Cech Jr. 1997; Wilkens et al. 2015), increased ventilation rates, and decreased metabolism (Secor and Niklitschek 2001). Low DO levels can reduce growth, feeding, and metabolic rates. Fish may swim to the surface in low oxygen conditions to receive more oxygen- rich water at the air-water interface (Shortnose Sturgeon Biological Assessment 2010; (Secor and Niklitschek 2001)). Hence, even a minor decrease in DO from dredging or dredge-related activities during these times can be harmful or fatal to sturgeon in rivers. This is particularly relevant when the dredged sediment contains high concentrations of organic material, similar to what is commonly seen in rivers of Southeast. These sediments often have high oxygen demands, and will actively absorb DO from the water column, lowering the oxygen available for other aquatic life.[26] Dredging these sediments can expose them to the water column where they can further degrade water quality beyond the changes in DO from dredging other types of sediments.

During times when DO is low, sturgeon may seek refuge from stressful environmental conditions by "hunkering" down and aggregating in deep, cool holes (Collins et al. 2002). The NMFS Greater Atlantic Region paper (NMFS 2020b) also stated that their important habitats for sturgeon include those suitable for resting, feeding, and aggregation areas that are attractive to sturgeon due to the physical characteristics of the river such as depth. For shortnose sturgeon in the Delaware River, these areas usually occurred in deeper waters (Hastings et al. 1987), which are similar to the seasonal aggregation areas identified in this Opinion in the Sturgeon PDCs in Appendix E. Additionally, sturgeon seek refuge from unsuitable water quality conditions (e.g., extreme temperatures and salinities) and during these times can tightly aggregate in relatively small areas within a river (e.g., a section that was less than 1 km in length) (Collins et al. 2002). Juvenile shortnose sturgeon that were tracked in the Savannah River traveled upriver when temperatures became too warm and downriver when the river temperatures were cooler (Collins et al. 2002).

When sturgeon aggregate in a particular location, there is an increased risk of take via direct interaction with dredge equipment. In addition, if they are aggregating in an area to seek refuge from stressful water quality conditions, dredging or dredge-related effects that force sturgeon to move from the area of refuge to a location that cannot support their physiological needs can also be harmful or fatal.

To protect sturgeon, the Sturgeon PDCs impose specific dredge restrictions based on the likely water quality conditions in a given river and the likelihood, or known existence, of a sturgeon seasonal aggregation. We developed a letter classification (A-D) system to denote the specific times and associated restrictions:

- Rivers identified with the letter "A" in of Appendix E refer to those with no known seasonal aggregations and historically good water quality. Rivers or stretches of river with this designation have no seasonal dredging restrictions because we anticipate water quality there is such that sturgeon would not aggregate in search of environmental refuge. Additionally, because the water quality is good, we do not anticipate dredging or dredge-induced effects would force animals from into an area that may not be able to support their physiological needs.
- For "B" rivers, seasonal aggregations have not been identified or no data on seasonal aggregations are available. However, all "B" rivers share a history of poor water quality during at least some portion of the year. The restrictions in these rivers are only limited to monitoring cutterhead dredging. Because the water quality in these rivers is poor, sturgeon residing here may be physiologically stressed and unable to avoid equipment

[26] Sediment oxygen demand (SOD) is the rate at which dissolved oxygen is removed from the water column during the decomposition of organic matter in streambed or lakebed sediments. In lakes and slow moving rivers, or rivers with high levels of organic matter in the bed sediment, SOD can be a major cause of low DO concentrations in the water column (https://pubs.usgs.gov/sir/2005/5228/; queried 11/18/2019).

that they normally could avoid during times of good water quality. As such, the potential for take during cutterhead dredging exists and must be monitored. A further discussion of potential cutterhead impacts to sturgeon in rivers is discussed in Section 6.1.3 of this Opinion.

- Designated "C" rivers are those rivers in which a seasonal aggregation has been identified. The aggregation areas are identified in Table 56 of Appendix E and include a buffer to ensure that dredge equipment does not directly contact sturgeon. Likewise, the buffer ensures turbidity and any associated decreases in DO generated by dredging occurring adjacent to the seasonal aggregation area does not reach the core aggregation area and potentially degrade the water quality in the aggregation area and force sturgeon to move from it.
- Rivers designated with a letter "D" designation refer to those for which no seasonal aggregation data and no water quality data are available. Since seasonal aggregations could occur in these rivers and/or water quality may be stressful, no dredging of any kind is allowed in these rivers during months when water quality in the Southeast is historically bad (June 1-September 30).

When the requirements associated with each of these categories and the other PDCs is Appendix E are properly implemented, we anticipate there will be no or insignificant effects from dredging in/around seasonal aggregations as a result of water quality changes, as discussed further below.

3.1.1.3 Mechanical Dredging (used for dredging and placement) and Geotechnical Surveys

Mechanical dredging equipment used for both dredging and placement of materials, and geotechnical surveys, may cause physical injury or mortality to ESA-listed species by striking animals with equipment or placement of material. Historically, NMFS has considered mechanical equipment used for dredging and material placement to be extremely unlikely to result in physical injury or other take of mobile ESA-listed species. Mechanical dredges are generally barge-mounted equipment that work in a set position before moving to the next location. Geotechnical surveys are conducted in generally the same way where equipment is lowered to take a core sediment sample; however, geotechnical surveys are a onetime sample collected from typically a 4-inch pipe and are therefore smaller in scope and scale than mechanical dredging.

For this Opinion, we reviewed available reported takes associated with mechanical dredging in the northeast and southeast U.S. and material placement equipment types provided in Table 10 to evaluate if we continue to support the determination that the likelihood of take associated with mechanical dredging and material placement is still extremely low. Reported take of ESA-listed species from mechanical dredging and material placement in the northeast and southeast U.S. is listed below:

Table 10. Reported Take Associated with Mechanical Dredging and Placement Equipment

Species	Date	Location	Information available on the reported take
Atlantic sturgeon	About 1998	Wilmington Harbor, North Carolina	Reported in NMFS Shortnose Recovery Plan (NMFS 1998) on page 53. No information available.
Shortnose sturgeon	6/19/2001	Kennebec River Bath Iron Works	Put in scow, released unharmed (Dickerson 2013)
Atlantic sturgeon	12/3/2002	Wilmington Harbor, North Carolina	Retrieved a 82.5 centimeter (cm), whole fish decomposing (Dickerson 2013)
Shortnose sturgeon	4/30/2003	Kennebec River Bath Iron Works	Fish nearly cut in half (Dickerson 2013)
Sea Turtle-species unknown	3/2/2005	Cape Canaveral, Florida	A sea turtle was killed by falling riprap being unloaded by a dump truck with a backhoe working to stop rocks from falling into the water. A copy of an undated letter in which Tiny's International Marine Environmental Services documented the event was provided to NMFS by the Canaveral Port Authority.
Green sea turtle	8/8/19	Cape Canaveral, Florida	A green sea turtle was observed 200 ft from a bucket dredge that was floating and unable to obtain proper buoyance to breathe. It was not recovered to determine the cause of the injuries.

3.1.1.3.1 Physical Injury to Sea turtles, Nassau Grouper, Elasmobranchs, Whales, and Sturgeon

We believe physical injury or other take of sea turtles (green, Kemp's ridley, hawksbill, leatherback, and loggerhead), sturgeon, Nassau grouper, elasmobranchs (giant manta ray, scalloped hammerhead, and smalltooth sawfish), and whales (blue, fin, sei, sperm, and North Atlantic right), by mechanical dredging is extremely unlikely to occur. We therefore believe that this route of effect is discountable for the following reasons.

We discovered reports of take of sea turtles and sturgeon (analyzed separately below) associated with mechanical dredging. The 2 reported sea turtle takes associated with mechanical dredging/placement equipment occurred in Cape Canaveral (Table 10). One apparently resulted from falling riprap, rather than dredge equipment, and the cause of the injuries in the 2019 report cannot be verified as related to dredging or other equipment used on the project. We have no information regarding any other reported takes caused by mechanical dredging equipment. Based on the circumstances of each take discussed above, and the infrequency with which those takes occurred relative to the overall amount of dredging, particularly within the action area, we still conclude that it is extremely unlikely that sea turtles, elasmobranchs (giant manta ray, smalltooth sawfish, or scalloped hammerhead shark), whales (blue, fin, sei, sperm, or North Atlantic right whale) would be injured by mechanical equipment, such as clamshell and bucket dredges used for dredging and material placement. Accordingly, these reported takes do not change our determination regarding the likelihood of mechanical dredging interactions with sea turtles, Nassau grouper, elasmobranchs, and whales. Mechanical equipment is generally stationary working from land or a barge and uses the bucket to either remove material when dredging or to place material. This type of equipment is extremely unlikely to move into a location where an ESA-listed species is positioned and encounter a mobile species without that species detecting its presence. Mobile ESA-listed species are expected to be able to avoid interaction with this slow process, even if they remain in the area (as sturgeon may do). In addition, the general PDCs require that crew members be aware of the species that could occur in the work area and monitor for their presence (General PDCs Section 2.1 of in Appendix B). If ESA-listed species are spotted within the distances provided in the PSO PDCs Section 1 of Appendix H, activities may not resume until the protected species has departed the project area of its own volition. The 2020 SARBO will also establish improved reporting of information from all dredging equipment types (e.g., other than just hopper dredging) that will allow us to continue to monitor if our determination remains accurate.

As discussed in Section 3.1.1.1 of this Opinion, we have information suggesting that sturgeon may not vacate areas where dredging is occurring as previously assumed. A study completed by the USACE ERDC evaluated the sturgeon takes reported to the USACE in the Atlantic from 1995-2013, and reported 3 sturgeon takes associated with mechanical dredging in rivers, shown in Table 10. Two of the reported sturgeon takes occurred at Kennebec River Bath Iron Works in an area where sturgeon were aggregating, which is believed to be the reason for the take. The Sturgeon PDCs (Appendix E) contain dredging restrictions that prohibit dredging in known sturgeon aggregation areas or where information regarding aggregations is unknown. Therefore, we do not expect that dredging in an aggregation area would occur as a result of the proposed action. The third reported sturgeon take in the ERDC study occurred in Wilmington Harbor in the Southeast region. The reported take was the capture of a decomposing fish during clamshell dredging, indicating the take may not have been associated with the clamshell dredge. Thus it is inappropriate to attribute to the risk of take from mechanical dredging under this Opinion. The fourth sturgeon take that occurred sometime around 1998, was reported in the NMFS Shortnose Recovery Plan (NMFS 1998) but no other information is available on this reported take. No other clamshell sturgeon takes are known to have occurred since 2013. We believe that physical injury to Atlantic and shortnose sturgeon as a result of mechanical dredging or geotechnical survey is extremely unlikely to occur, based on the rarity of interactions between mechanical dredging equipment and sturgeon, and the protective measures contained in the PDCs, as discussed above, to avoid mechanical dredging in aggregation areas.

3.1.1.3.2 Water Quality Impacts to Sea turtles, Nassau Grouper, Elasmobranchs, Whales, and Sturgeon

We believe changes in water quality resulting from turbidity from mechanical dredging and material placement analyzed under this Opinion may affect, but are not likely to adversely affect sea turtles (green, Kemp’s ridley, hawksbill, leatherback, and loggerhead), sturgeon, Nassau grouper, elasmobranchs (giant manta ray, scalloped hammerhead, and smalltooth sawfish), and whales (blue, fin, sei, sperm, and North Atlantic right), for the following reasons.

Mechanical dredging that scoops material and pulls it through the water column is expected to create turbidity plume causing a decrease in the near field DO concentration. We believe any potential exposure to temporary turbidity and the resulting sedimentation generated by mechanical dredging and material placement covered under this Opinion will have an insignificant effect on mobile ESA-listed species, particularly outside of riverine environments, as they have unrestricted access to be able to move away from the turbidity generated, and to continue to use similar habitat nearby, if needed. As discussed above, a notable exception may be sturgeon in rivers, they may not be able to or may not elect to avoid these areas. Open water environments such as estuaries and open ocean areas in the action area are expected to have adequate water flow to ensure good water quality including sufficient DO for mobile species year round. The General PDCs in Section 2.2 of Appendix B, require that material and equipment be placed in a manner that will not block the movement of species in the area and therefore these species will be able to move around and avoid localized areas of turbidity in open water environment (e.g., turbidity curtains will not block species from entering or leaving an area). In addition:

- Sea turtles: Turbidity is not generally believed to impact sea turtles, as sea turtles breathe air and can therefore both move away from areas of poor water quality and surface to breathe air.
- Sturgeon in open water areas and Nassau Grouper: Studies of the effects of turbid water on fishes suggest that concentrations of suspended solids can reach thousands of milligrams per liter (mg/L) before an acute toxic reaction is expected (Burton 1993). Any turbidity exceeding those thresholds under this Opinion would be localized to the project location. Fish such as the Nassau grouper and sturgeon will be able to avoid localized areas of turbidity in open water environments, if needed. Additionally, we expect any turbidity will be temporary, lasting only for the duration of the proposed project. We therefore expect that these fishes in open water environments will not be exposed to harmful levels of turbidity.
- Sturgeon in riverine environments: As noted in Section 3.1.1.2 of this Opinion, during periods of stressful water quality (primarily summer months) even small decreases in DO can harm sturgeon. The PDCs establish a 3,000 ft buffer zones around the known seasonal aggregations areas identified in the Sturgeon PDCs in Appendix E to protect sturgeon from stressful decreases in DO. This distance is the furthest downstream Burton (Burton 1993) measured total suspended solid concentrations from dredge sites in the Delaware River. We believe buffer zones of this size are sufficiently large to ensure the turbidity, and resultant changes in DO concentrations, we anticipate would be cause by any form of mechanical dredging will have dissipated before reaching sturgeon within the aggregations. Thus, we anticipate any adverse effects would be insignificant.
- Elasmobranchs: Smalltooth sawfish are often exposed to more turbid waters in estuarine environments and therefore are not expected to be stressed by exposure to turbidity. Additionally, all elasmobranch species can avoid localized areas of increased turbidity, if needed.
- Whales: Whales are not expected to impacted by turbidity, as whales breathe air and can therefore both move away from areas of poor water quality and surface to breathe air. In addition, blue, fin, sei, and sperm whales occur in deeper water environments away from most activities covered under except placement in ODMDS and borrow area dredging. The PDCs require that all work cease if whales are spotted in the area.

We believe that geotechnical surveying would have similar effects as mechanical dredging in that equipment is placed in the water to collect sediment, but note that taking a single geotechnical sample with typically a 4-inch pipe will be less of a risk than continuing to mechanical dredge an area over a period of time. We have no reports of interactions between ESA-listed species from geotechnical surveys. We also have no reason to expect that geotechnical surveys would impact water quality. Therefore, we similarly believe physical injury or other take of ESA-listed species by geotechnical surveying is extremely unlikely to occur, and therefore that this route of effect is also discountable.

3.1.1.4 Cutterhead Dredging

Historically, we have considered the risk of a mobile species encountering a cutterhead dredge to be extremely low, such that this route of effect is discountable. NMFS reviewed all available reports of take in the Northeast and Southeast associated with cutterhead dredging to evaluate if that determination is still accurate. This review of reported takes included reports of sea turtles and sturgeon takes associated with cutterhead dredging (Table 11), which are evaluated further below in Section 3.1.1.4.1 of this Opinion for sea turtles and Section 3.1.1.4.2 of this Opinion for Atlantic and shortnose sturgeon. We did not find any reported takes associated with cutterhead dredging of any other ESA-listed species in the action area including: Nassau grouper, elasmobranchs (giant manta ray, smalltooth sawfish, or scalloped hammerhead shark), or whales (blue, fin, sei, sperm, or North Atlantic right whale). We also have no reports of takes of sturgeon associated with cutterhead dredging in open water environments. We believe there will be no effect to elasmobranchs or sturgeon in open water areas, and whales by physical injury from cutterhead dredging since they are highly mobile and can avoid interactions with both the equipment and the suction created by cutterhead dredging.

Cutterhead dredges are a suction type dredge that operate when the cutterhead is generally embedded in sediment, which is also a PDC requirement for projects covered under this Opinion (Section 3.3 of the General PDCs in Appendix B). The cutterhead creates a small zone of suction around the cutterhead, which if the cutterhead were to be exposed to the water column when not completely embedded in sediment could expose species to the risk of entrainment. We believe smaller fish who are not strong enough to outswim the suction zone or larger individuals who are biologically motivated to remain in place may not swim away from equipment and could be injured. Of the species analyzed in this Opinion, only smaller individuals or larger individuals biologically motivated to remain in place would be at an increased risk of entrainment. This includes Atlantic and shortnose sturgeon in spawning rivers (analyzed in Section 3.1.1.4.2 of this Opinion) and Nassau grouper in the U.S. Caribbean (analyzed in Section 3.1.1.4.3 of this Opinion).

Cutterhead dredging may cause localized turbidity; however, we expect that in open water environments mobile species will avoid these disturbed areas if needed and turbidity will dissipate relatively quickly. Therefore, for the same reasons discussed in Section 3.1.1.3.1 for mechanical dredging, we expect any effects to sea turtles, sturgeon in open water areas, Nassau grouper, elasmobranchs or whales as a result of changes in water quality from cutterhead dredging to be insignificant. Potential effects to sturgeon in riverine environments are discussed specifically at Section 3.1.1.4.2 of this Opinion.

Table 11. Reported Take Associated with Cutterhead Dredging

Date	Location	Species	Dredge Type/ Name	Status	Species Details	Data Source
1992	Columbia River, Oregon	Sturgeon	Cutterhead	Dead	Substantial number of 300-500 millimeters (mm) sturgeon	(Reine et al. 2014)
2/1/96	Delaware River (Newbold Island)	shortnose sturgeon	cutterhead *Ozark*	Dead	83 cm, female w/eggs	(Reine et al. 2014)
2/1/96	Delaware River (Newbold Island)	shortnose sturgeon	cutterhead *Ozark*	Dead	63 cm, mature male	(NMFS 2017a)
1/6/98	Delaware River (Kinkora Range)	shortnose sturgeon	Assumed cutterhead	Dead	Either 657 mm or 573 mm	(NMFS 2017a)
1/12/98	Delaware River (Florence Range)	shortnose sturgeon	Assumed cutterhead	Dead	Either 657 mm or 573 mm	(NMFS 2017a)
1/13/98	Delaware River (Florence Range)	shortnose sturgeon	Assumed cutterhead	Dead	Either 657 mm or 573 mm	(NMFS 2017a)
unknown	Wilmington Harbor, Cape Fear River, North Carolina	Atlantic sturgeon	Assumed cutterhead	Dead	none provided	NMFS 1998 Shortnose Recovery Plan p. 53
12/24/04	Brownsville Entrance Channel, Texas	green sea turtle	cutterhead	Alive, cracked plastron and carapace	none provided	2008 SARBA

3.1.1.4.1 Sea Turtles

Potential effects to sea turtles by cutterhead dredging include physical injury. We believe this route of effect is discountable. Information provided by the USACE in the 2008 SARBA reported that the 1 documented sea turtle interaction with a cutterhead dredge was based on a live stranded green sea turtle discovered outside of the dredge discharge area with a cracked plastron and carapace. This stranding was 1 of 42 cold-stunned green sea turtle strandings in the area during a cold front that swept through South Texas on December 22, 2004, and therefore cannot be definitively linked to injury caused by the cutterhead dredge. NMFS has no other information or reported takes of sea turtles by cutterhead dredging, despite frequent use of cutterhead dredging within the action area. Therefore, we believe the risk of physical injury or take of sea turtles (green, Kemp’s ridley, hawksbill, leatherback, and loggerhead) by cutterhead dredging is an extremely unlikely event that we do not expect occur. We continue to expect that sea turtles will move away from and avoid interaction with cutterhead dredging. Further, the risk assessment plan process (Section 2.9 of this Opinion) will consider project timing factors that may affect sea turtles’ ability to avoid cutterhead dredges, such as working during times of year when fewer sea turtles may be present in an area, and avoiding times when water temperatures drop too low, as the risk of take of cold-stunned sea turtles may increase. Understanding and adjusting work to further minimize risk through the risk assessment process provides an additional tool to ensure that the likelihood of a sea turtle encountering a cutterhead dredge remains extremely low.

3.1.1.4.2 Sturgeon in Rivers

When evaluating the risk of sturgeon take by cutterhead dredging in riverine environments, we consider the likelihood of sturgeon to avoid dredging and the ability for sturgeon of varying size classes to be able to outswim the suction generated by cutterhead dredging, as previously discussed. As discussed in Section 3.1.1.1of this Opinion, sturgeon may not avoid dredging or the equipment associated with it. Thus, there is nothing inherent about the operation of cutterhead dredging that would appear to "repel" sturgeon. Instead, it appears the likelihood of take occurring is largely a function of: (1) sturgeon proximity to the dredge head (i.e., sturgeon density) and (2) an individual sturgeon's ability to escape/avoid the dredge head if nearby.

The cutterhead dredge operates with the dredge head buried in the sediment; however, a flow field is produced by the suction of the operating dredge head. The amount of suction produced is dependent on linear flow rates inside the pipe and the pipe diameter (Clausner and Jones 2004). High flow rates and larger pipes create greater suction velocities and wider flow fields. The suction produced decreases exponentially with distance from the dredge head (Boysen and Hoover 2009).

To specifically evaluate the behavior of sturgeon around cutterhead dredges Environmental Research and Consulting (ERC 2012) tracked movements of tagged Atlantic and shortnose sturgeon around active cutterhead dredge sites in riverine environments. Nineteen tagged Atlantic sturgeon and 3 tagged shortnose sturgeon (all juveniles) were in the study area during the time dredging was ongoing. Eleven of the 19 juvenile Atlantic sturgeon detected during this study remained upriver of the dredging area. Three of the juvenile sturgeon detected during this study (2 Atlantic sturgeons and 1 shortnose sturgeon) appeared to have moved through the study area when the dredge was working. The patterns and rates of movement of these fish indicated nothing to suggest that their behavior was affected by dredge operation. The remaining sturgeon either moved through the study area before or after the dredging was happening. It is unknown whether some of these fish chose behaviors (routes or timing of movement) that kept them from the immediate vicinity of the operating dredge (ERC 2012).

A similar study was carried out in 2009 in the James River (Virginia) (Cameron 2012). Six subadult Atlantic sturgeon (77.5 – 100 cm length) were caught, tagged with passive and active acoustic tags, and released at the dredge site. The study concluded that: tagged fish showed no signs of impeded up- or downriver movement due to the physical presence of the dredge; fish were actively tracked freely moving past the dredge during full production mode; fish showed no signs of avoidance response (e.g., due to noise generated by the dredge) as indicated by the amount of time spent in close proximity to the dredge after release (3.5 – 21.5 hours); and, tagged fish showed no evidence of attraction to the dredge (Cameron 2012).

Aside from observing the behavioral responses of sturgeon to cutterhead dredges, several studies have also attempted to understand the physical ability of sturgeon to avoid cutterhead dredges. A study in 2011 (Hoover et al. 2011) demonstrated the swimming performance of juvenile lake sturgeon and pallid sturgeon (12 – 17.3 cm fork length) in laboratory evaluations. The authors compared swimming behaviors and abilities in water velocities ranging from 10 to 90 centimeter per second (0.33-3.0 feet per second). At distances more than 1.5m from the dredges, water velocities were negligible (10 centimeter per second). The authors conclude that for a sturgeon to be entrained in a dredge, the fish would need to be almost on top of the drag head and be unaffected by associated disturbance (e.g., turbidity and noise) (Hoover et al. 2011). The authors also conclude that juvenile sturgeon are only at risk of entrainment in a cutterhead dredge if they are in close proximity, less than 1 m, to the dragheads (Hoover et al. 2011).

Boysen and Hoover (2009) assessed the probability of entrainment of juvenile white sturgeon by cutterhead dredge by evaluating swimming performance of young–of-the-year fish (8-10 cm total length). The authors determined that within 1 m of an operating dredge head, all fish would escape when the pipe was 61 cm (2 ft) or smaller. Fish larger than 9.3 cm (about 4 in) would be able to avoid the intake when the pipe was as large as 66 cm (2.2 ft). The authors concluded that regardless of fish size or pipe size, fish are only at risk of entrainment within a radius of 1.5 – 2 m of the dredge head; beyond that distance velocities decrease to less than 1 feet per second.

Clarke et al. (2011) reports that a cutterhead dredge with a suction pipe diameter of 36 (Clarke et al. 2011)inches has an intake velocity of approximately 95 centimeter per second at a distance of 1m from the dredge head and that the velocity reduces to approximately 40 centimeter per second at a distance of 1.5 m, 25 centimeter per second at a distance of 2 m and less than 10 centimeter per second at a distance of 3 m. Clarke also reports on swim tunnel performance tests conducted on juvenile and subadult Atlantic, white, and lake sturgeon. He concludes that there is a risk of sturgeon entrainment only within 1m of a cutterhead dredge head with a 36-inch pipe diameter and suction of 4.6 meter per second.

The risk of an individual sturgeon being entrained in a cutterhead dredge is difficult to calculate. While a large area overall will be dredged, the dredge operates in an extremely small area at any given time (i.e., the river bottom in the immediate vicinity of the intake). To be entrained, an individual would need to be in the immediate area where the dredge is operating (i.e., within 1m of the dredge head). It is likely that nearly all shortnose and Atlantic sturgeon in the action area will never encounter the dredge as they would not occur within 1 m of the dredge.

However, Reine et al. (2014) evaluated the sturgeon takes reported to the USACE from 1995-2013 that were associated with cutterhead dredging, including 5 reports of shortnose sturgeon found in disposal sites of an area being dredged in the Delaware River by cutterhead in the NMFS Greater Atlantic Region Table 11. Additionally, with a cutterhead dredge, material is pumped directly from the dredged area to a disposal site. As such, there is no opportunity to monitor for biological material on board the dredge; rather, observers work at the disposal site to inspect material. NMFS Greater Atlantic Region reports that all 5 cutterhead dredge takes described in Reine et al. (2014) occurred in known overwintering aggregation areas, where "shortnose sturgeon rest on the bottom and exhibit little movement and may be slow to respond to stimuli such as an oncoming dredge" (James River Federal Navigation Project, NMFS tracking number NER-2018-15090). We believe sturgeon in the Southeast exhibit similar "hunkering" behavior in certain rivers during summer months when water temperatures are high and DO concentrations are low, as discussed in Section 3.1.1.2. We believe dredging during times when water quality is poor and sturgeon are stressed, that they are at an increased risk of entrainment in cutterhead dredging, similar to what occurred in the Delaware River. To minimize this risk to sturgeon, the Sturgeon PDCs prohibit dredging in known sturgeon seasonal aggregation areas and require monitoring of cutterhead dredging outside of aggregation areas in the sections of sturgeon rivers identified as having poor water quality (identified as sections and times with the letters "B" or "C" Table 56 in the Sturgeon PDCs in Appendix E. We therefore believe that take of Atlantic and shortnose sturgeon will occur by cutterhead dredging in rivers during the times identified as "B" or "C" Table 56 in the Sturgeon PDCs in Appendix E, which is discussed further in Section 6.1.3 of this Opinion.

Cutterhead dredging removes sediment by suction and, as required by the PDCs, is not operated until the dredging cutterhead is embedded in the sediment. While they may create a small turbidity plume localized around the dredging head, this plume is expected to be localized and changes in DO would also be expected to be minimal. A recent study (USACE 2019b) measuring changes in DO around a cutterhead dredge in the Savannah River noted that the greatest change in DO occurred in the bottom third of the water column where the cutterhead was working. Changes in DO in the bottom of the water column were most notable within 50 m downstream of the dredge and returned to background levels within 100 m of the dredge with all changes occurring directly downstream and did not extend the width of the river. (ERDC 2019), measured DO both up and downstream of the cutterhead dredge over multiple days, in multiple locations, using continuous monitoring and handheld equipment. The greatest change measured was from continuous monitoring that showed an average of 0.4 mg/L drop 50 m downstream of the dredge in the bottom of the water column. Specifically, the downstream DO average was 2.7 mg/L (minimum = 1.9, maximum = 3.1) and the upstream average DO was 3.1 mg/L (minimum = 2.3, maximum = 3.3). This minor drop in DO is likely due to the suction nature of cutterhead dredges, which minimize the turbidity plume. Cutterhead dredges also pump water from near the water's surface to the cutterhead blade to assist with dredging. This action draws in at least some water from the surface that is expected to be more oxygen rich and moves it to the sea floor where DO levels are typically lower. ERDC (2019) also reported that the cutterhead dredge "Hampton Roads" pumped 480-700 gallons per minute of water from 0.7 m depth down to the cutterhead operating at the bottom of the river. Because of the very small area where cutterhead dredging is removing sediment once embedded in the sediment, turbidity generated and the area of lower DO is localized and returns to normal quickly in riverine environments due to the water flow and is expected to have an insignificant effect to sturgeon in rivers, outside of seasonal aggregation areas, even during times of poor water quality. For animals inside the seasonal aggregation areas, we anticipate the buffer zones established in the Sturgeon PDCs are sufficiently large to ensure the turbidity, and resultant changes in DO concentrations, caused by cutterhead dredging will have dissipated before reaching sturgeon within the aggregations. Thus, we anticipate any adverse effects to animals inside the seasonal aggregation areas from water quality changes caused by dredging will also be insignificant.

If new seasonal aggregation areas are discovered, they will be added to the exclusion areas in the tables identifying seasonal aggregation areas and upper river limits for work in the Sturgeon PDCs during the annual review (Section 2.9.4.1 of this Opinion). If new spawning areas are discovered, they will be added to the upper river limits for work in the Sturgeon PDCs during the annual review (Section 2.9.4.1 of this Opinion) to ensure this Opinion is protective of smaller fish. This Opinion also limits work in sturgeon rivers to areas where smaller larval stages or sturgeon are not expected to occur, so injury of these smaller fish is not expected.

3.1.1.4.3 Nassau Grouper

As stated above, we believe that only larval and small juvenile Nassau grouper may be small enough to be unable to outswim the suction generated by a cutterhead dredging. Nassau grouper is a demersal (bottom-dwelling) fish that associates with habitat types where work will not be occurring under this Opinion such as hardbottom, reef, and other hard structures in South Florida and nearshore lagoon habitat with seagrass and mangrove habitat in the U.S. Caribbean. The General PDCs in Section 2.2 of Appendix B limit the proximity and duration of work on or near hardbottom, reef, and seagrass habitats. In addition, the Coral PDCs protect hardbottom and reef habitat within the range of ESA-listed corals which overlaps with the range of this species (Coral PDCs in Appendix C). Therefore, cutterhead dredging will not occur in areas where larval or small juvenile Nassau grouper are expected to be present. In the U.S. Caribbean and Florida Keys, Nassau grouper (19 cm fork length and larger) could occur over corals, reefs, and other hardbottom habitat, including channels and canals cut through the limestone hardbottom like those that where maintenance dredging covered under this Opinion can occur. However, Nassau grouper at this size are considered larger juveniles that are highly mobile and are expected to be large enough that they could outswim the suction generated by a cutterhead dredging and that the risk of entrainment would be extremely low, such that we consider this route of effect to be discountable.

3.1.1.5 Hopper Dredging

Hopper dredges are known to cause mortality to sea turtles (Green, Hawksbill, Kemp's ridley, and Loggerhead) and sturgeon (Atlantic and shortnose), based on monitoring for sea turtle takes since 1980, by entrainment and impingement. We therefore believe that hopper dredging is likely to continue to adversely affect these species, as described below and discussed in Section 6 of this Opinion. Species can become entrained in hopper dredges as the draghead moves along the bottom. Entrainment occurs when the species cannot escape from the suction of the dredge and they are sucked into the dredge draghead, pumped through the intake pipe, and then killed as they cycle through the centrifugal pump and into the hopper. Because entrainment is believed to occur primarily while the draghead is operating on the bottom, it is likely that only those species feeding or resting on or near the bottom would be vulnerable to entrainment. They can also be entrained if suction is created in the draghead by current flow while the device is being placed or removed, or if the dredge is operating on an uneven or rocky substrate and rises off the bottom. Recent information from the USACE suggests that the risk of entrainment is highest when the bottom terrain is uneven or when the dredge is conducting "cleanup" operations at the end of a dredge cycle when the bottom is trenched and the dredge is working to level out the bottom. In these instances, it is difficult for the dredge operator to keep the draghead buried in the sand, thus species near the bottom may be more vulnerable to entrainment. Sea turtles or sturgeon resting in deeper waters or holes in the channel may be at an increased risk of take from dredging activities conducted there. Species can also be crushed on the bottom by the moving draghead and not entrained.

The USACE has been hopper dredging, and authorizing hopper dredging since the 1980s. However, the issuance of the 1997 SARBO resulted in significant changes to dredging practice, such as the requirement to have a PSO present and the use of draghead deflectors during hopper dredging. These protective measures resulted in reduced take from hopper dredging activities; therefore, hopper dredging take reported before the 1997 SARBO is not representative of take expected under this Opinion and is not used for estimating future take. We reviewed all takes that occurred in the 21 years since the issuance of the 1997 SARBO (fiscal year 1998-2018, which is the last complete year of data available), as shown in Table 12, Reported Takes from Hopper Dredging per year since 1997 SARBO. All hopper dredging take associated with the 1997 SARBO and other major USACE dredging projects are documented on the ODESS from 1998 to present.

Table 12. Reported Takes from Hopper Dredging per year since 1997 SARBO[27].

Fiscal Year	Green sea turtle	Kemp's ridley sea turtle	Loggerhead sea turtle	Unknown sea turtle	Atlantic sturgeon	Unknown sturgeon	Hopper Volume (cy)
1998	1	0	9	0	0	0	5,657,819
1999	1	1	13	0	1	0	6,253,794
2000	1	2	18	0	2	0	14,821,757
2001	2	0	3	0	0	0	2,908,339
2002	0	7	12	0	1	0	9,065,303
2003	0	0	5	0	0	0	4,816,289
2004	4	2	7	0	0	0	4,836,651
2005	0	0	6	0	2	0	11,867,599
2006	0	6	5	0	0	0	6,875,942
2007	1	1	7	0	1	0	7,640,337
2008	1	3	6	0	1	0	6,523,530
2009	5	0	7	0	0	0	14,382,100
2010	0	1	4	0	4	0	8,417,827
2011	9	0	12	0	1	0	6,987,091
2012	8	7	17	1	1	0	9,808,468
2013	2	1	6	0	0	0	7,362,809
2014	1	1	5	0	0	0	9,318,799
2015	1	3	10	0	2	1	7,120,000
2016	9	8	10	0	6	1	12,634,000
2017	22	16	28	0	17	1	10,417,000
2018	2	5	8	4	14	0	9,102,000
Total	**70**	**64**	**198**	**5**	**53**	**3**	**176,817,454**

[27] Note: Years that take of a specific species exceeded the annual number allowed under the 1997 SARBO (shaded grey in the table. Take reported as (+ number) represent channel deepening projects at Savannah, Jacksonville, and Charleston Harbor and Bogue Bank. These projects are in the action area and will continue to be maintained under this Opinion. Atlantic sturgeon were not listed under the ESA until 2012. Reports in this table prior to 2012 are for information only and to aid in future take estimates.

3.1.1.5.1 Sea Turtles and Sturgeon

To date, only green, Kemp's ridley, and loggerhead sea turtles and Atlantic and shortnose sturgeon have been reported as taken by hopper dredging in the South Atlantic, as shown in Table 12. As a result, we believe that there will be adverse effects to green (NA and SA DPSs), Kemp's ridley, and loggerhead sea turtles, and Atlantic and shortnose sturgeon from entrainment or impingement due to hopper dredging. The adverse effects of hopper dredging, anticipated take, and all minimization measures required in this Opinion are discussed further in Section 6 of this Opinion.

We believe hopper dredging will not affect leatherback or hawksbill sea turtles. There are no reports of take of leatherback or hawksbill sea turtles from hopper dredging in the action area. Hawksbill sea turtles are generally not vulnerable to entrainment due to their association with reef habitat where hopper dredging will not occur under this Opinion. Leatherback sea turtles are generally not vulnerable to entrainment due to their large size and generally pelagic habits. While there are 3 reported captures of leatherback sea turtles in 2017 and 2 in 2018 in the action area (Table 12), these reports were not considered to be a hopper dredge take. The 3 reported leatherback sea turtle takes in 2017 occurred in Brunswick Harbor and were observed in the hopper dredge already severely decomposed and assumed to be parts of potentially the same or 2 different turtles, which accordingly were not attributable to dredging. The 2 turtles were reported entangled in floating buoy lines associated with the project, but not a result of dredging. Based on the lack of reported interactions, and these species expected avoidance of hopper dredging activities, we believe that hopper dredging will have no effect on leatherback or hawksbill sea turtles.

No water quality effects that may adversely affect sturgeon or sea turtles are anticipated. Overflow from hopper dredging or from other equipment such as barges and scows could increase turbidity in the area, and would likely cause a decrease in DO concentrations. However, hopper dredging covered under this Opinion is limited by the PDCs to times of year in sturgeon rivers when water quality is not seasonally degraded in (e.g., winter). Additionally, as explained in Section 3.1.1.3.1, sturgeon and sea turtles will be able to avoid localized areas of turbidity in open water environments, if needed. Further, any turbidity will be temporary, lasting only for the duration of the proposed project. We therefore do not anticipate any adverse effects to sturgeon or sea turtles from changes in water quality or the associated decrease in DO concentration associated with these activities.

3.1.1.5.2 Nassau Grouper

We believe that entrainment and impingement from hopper dredging of Nassau grouper is extremely unlikely to occur. We therefore believe that this route of effect is discountable. Nassau grouper are not expected to be present in many areas where hopper dredging occurs under the proposed action. Larval and juvenile Nassau grouper are not expected to be encountered due to their association with nearshore lagoon habitat with seagrass and mangrove habitat in the U.S. Caribbean, because hopper dredging will not occur in this environments. The General PDCs in Section 2.2 of Appendix B limit work where adult Nassau grouper are expected to be present within portions of the action area, including on or near hardbottom, reef, and seagrass habitats. In addition, the Coral PDCs protect hardbottom and reef habitat within the range of ESA-listed corals which overlaps with the range of this species (Coral PDCs in

Appendix C). In the U.S. Caribbean and Florida Keys, Nassau grouper (19 cm fork length and larger) could be present in the navigation channels that are maintenance dredged, but we believe that any interaction is highly unlikely, and therefore believe that route of effect is discountable. We have no reports of take of Nassau grouper by hopper dredging and Nassau grouper (19 cm fork length and larger) are highly mobile and would be expected to avoid active construction equipment.

No water quality effects that may adversely affect Nassau grouper are anticipated. As explained in Section 3.1.1.3.1, fish such as the Nassau grouper and sturgeon will be able to avoid localized areas of turbidity in open water environments, if needed. Additionally, any turbidity will be temporary, lasting only for the duration of the proposed project. We therefore expect that any effects would be insignificant.

3.1.1.5.3 Elasmobranchs, and Whales

We believe that there is not a risk of entrainment and impingement from hopper dredging (no effect) to elasmobranchs (giant manta ray, smalltooth sawfish, or scalloped hammerhead shark), and whales (blue, fin, sei, sperm, or North Atlantic right whale) from hopper dredging. Based on NMFS' decades of experience with reporting of take from hopper dredging (since the 1980's), and a review of the available scientific literature, NMFS determined that to date, there have been no known reports of hopper dredging entrainment of these species. Elasmobranchs and whales are not expected to be entrained due to their large size and ability to avoid the suction created by a hopper dredge. In addition, blue, fin, sei, and sperm whales are expected to generally occur in deeper waters than where hopper dredging will occur, and the PDCs require that all work cease if whales are spotted in the area.

No water quality effects that may adversely affect elasmobranchs or whales are anticipated as a result of hopper dredging. As explained in Section 3.1.1.3.1, all elasmobranch species can avoid localized areas of increased turbidity, if needed, and whales breathe air and can therefore both move away from areas of poor water quality and surface to breathe air. In addition, blue, fin, sei, and sperm whales are generally located in deeper waters off the continental shelf and therefore away from most dredging activities borrow area dredging.

3.1.1.6 Agitation Dredging (bed-leveling and WID)

3.1.1.6.1 Sea Turtles and Sturgeon

Sea turtles (green, Kemp's ridley, hawksbill, leatherback, and loggerhead) and sturgeon (Atlantic and shortnose) may be injured or killed if struck by bed-leveling or WID equipment. However, we believe that the potential for physical impacts to species from bed-leveling and WID is so low that we consider this route of effect to be discountable. Due to uncertainty in the effects to sea turtles and sturgeon from bed-leveling, the USACE's Savannah District performed a study and published a report titled *Bed-Leveler Evaluation Report (USACE 2013)*. In this study, closed-net trawlers and observers were used prior and post bed-leveling for 2 years (2013 and 2014) to trawl around the area to determine if Atlantic sturgeon or sea turtles in the area were first present and second if they were injured. The results indicate that bed-leveling did not result in injury or death of either sea turtles or sturgeon, likely due to the slow speed of the equipment and the sand wedge created in front of the bed-leveler that prompts sea turtles and sturgeon to move off the channel bottom and away from the bed-leveler. Brunswick Harbor, Georgia, was chosen for this study based on the density of sea turtles in the area and the concern of sea turtle brumation (hibernating on the sea floor during cold weather events) leading to their being injured by bed-leveling. During the trials, trawling directly behind the bed-leveler captured and released 38 live sea turtles and 2 Atlantic sturgeon with no mortalities, thus demonstrating that sea turtles and sturgeon were present during the bed-leveling operations and unharmed by the process. WID, like bed-leveling, uses a slow moving device and both sea turtles and sturgeon are expected to move away from the injection head. Both Atlantic sturgeon and shortnose sturgeon are expected to be capable of swimming speeds greater than those at which bed-leveling and WID equipment is towed (1-2 knots).

The PDCs of this Opinion allow the use of bed-leveling designs not specifically considered in the Savannah District study. However, we expect the effects from any alternative designs to be the same as those previously tested, as the bed-leveling PDCs in the General PDCs Section 3.4 of Appendix B requires that all designs meet the same objective of creating a disturbance ahead of the equipment, which is understood to cause animals to move away from the equipment, and prohibits designs with areas on the bed-leveler that could create a pinch point and trap ESA-listed species. In addition, the bed-leveling PDCs require that the local sea turtle stranding network be alerted if any dredging is occurring in their area and particularly if bed-leveling is occurring so they can monitor for strandings that may be associated with these new bed-leveling designs. Any shift in effects observed from bed-leveling will be reviewed as part of the programmatic annual review in Section 2.9.2 of this Opinion. If it is determined that bed-leveling techniques are resulting in effects not considered in this Opinion, reinitiation may be triggered.

3.1.1.6.2 Sturgeon in Rivers

In addition to the potential for physical injury discussed above, sturgeon in rivers are particularly susceptible to changes in water quality from activities associated with this Opinion. As discussed in Section 3.1.1.3.2, other mobile ESA-listed species considered in this Opinion are significantly less vulnerable to changes in water quality, and we do not expect that these species will be affected by changes in water quality associated with bed-leveling. Bed-leveling is frequently used in sturgeon rivers to move sediment from an area where it is accumulating, such as a berth, back into the river to be washed out of the area by the river water movement and often by the tide in areas of the river closer to the ocean. For instance, bed-leveling, has been used to dredge Savannah Harbor since 1932 (Way et al. 2007). A study of the effects of bed-leveling on water quality in Savannah Harbor was completed in the spring and fall of 2000 (Way et al. 2007) that reported suspended material was generally limited to the bottom 3 m of the river and extended 2,000 ft. downstream with the highest concentrations of turbidity limited to just downstream of the tug pulling the bed-leveler. This study noted a number of factors that can affect the extent of turbidity generated and the changes in DO resulting from bed-leveling. The factors include: the tide cycle (ebb, flood, or flow tides), the velocity of the water, amount of fines in the sediment moved with finer grain sediments traveling further, and the type of material being dredged. Way (2007) monitored water quality effects from both overboard in-water disposal and bed-leveling and did not directly measure changes in DO from bed-leveling, using in-water disposal as a proxy. It found some sediments in the Savannah Harbor contained decomposing organic material trapped that lowered DO when agitated into the water column.

In-water disposal of this material resulted in a maximum drop of 2.8 mg/L within 100 ft. downstream during a spring disposal event, which was 1 mg/L lower than background levels 1,000 ft. downstream. The *Biological Assessment for Agitation and Water Injection Dredging Port of Morehead City, North Carolina State Ports Authority* (NCSPA 2017) reported the worst-case scenario observed in the Savannah River was an increase of TSS of over 300 mg/L extending 1,500 ft. downstream and 400 ft. across the width of the river, indicating that the turbidity and likely lowering of DO from bed-leveling can affect a substantial area while operating. Another study in 1975 (Hussey et al. 1975), resulted in the same conclusion after monitoring agitation dredging in Savannah Harbor concluding, "DO was reduced either not at all or by small amounts. The latter situation became a potential problem on 2 occasions when background DO levels were very low and agitation dredging reduced them to less than acceptable values."

Water-Injection Dredging (WID), as described in Section 2.5.3.1.2, is another form of agitation dredging similar to bed-leveling. WID uses high volumes of low pressure water, pumped through a series of nozzles on a wide horizontal jetbar directly into the bottom sediments. Injecting water creates a very fluid mud layer that remains close to the bottom and is washed away by the outgoing tide until settling out further downstream. WID is used to move relatively thin layers of sediment (typically less than 1 ft. deep). Multiple studies have shown that in rivers, this liquid mud layer remains in the bottom 2 ft. of the water column and settles out within 1,000-2,000 ft. downstream. A study (Law Engineering and Environmental Services 1998) using this method in the Port of Wilmington found that while it increased turbidity, which can result in lower DO, it did not result in a statistically significant change in DO after WID compared to baseline conditions, even during periods of already decreased DO.

Law (1998) was referenced in the *Biological Assessment for Agitation and Water Injection Dredging Port of Morehead City, North Carolina State Ports Authority (NCSPA 2017)*, which concluded that the vertical and horizontal mixing of bottom anoxic water during agitation dredging likely raises the DO level within the plume, thereby maintaining DO levels. We believe it is logical that DO changes would be minor in most cases using WID, but even minor changes could lower the DO levels to be harmful to sturgeon, especially if the sediment transported is high in organic material or areas where contaminants could further deplete DO concentrations.

As noted in Section 3.1.1.2 of this Opinion, during periods of stressful water quality (primarily summer months) even small decreases in DO can harm sturgeon, which is why buffer zones were established around the known seasonal aggregations areas identified in the Sturgeon PDCs in Appendix E. We have specifically implemented buffer zones we believe are sufficiently large to ensure the turbidity, and resultant changes in DO concentrations, associated with any form of agitation dredging will have dissipated before reaching any sturgeon within the aggregations. Thus, we anticipate any adverse effects would be insignificant.

3.1.1.6.3 Nassau Grouper

We believe that there will be no effect to Nassau grouper from agitation dredging. Nassau grouper is a demersal (bottom-dwelling) fish that associates with habitat types where work will not be occurring under this Opinion such as hardbottom, reef, and other hard structures in South

Florida and nearshore lagoon habitat with seagrass and mangrove habitat in the U.S. Caribbean. The General PDCs in Section 2.2 of Appendix B limit work on or near hardbottom, reef, and seagrass habitats. In addition, the Coral PDCs protect hardbottom and reef habitat within the range of ESA-listed corals which overlaps with the range of this species and do not allow the use of agitation dredging in these areas (Coral PDCs in Appendix C).

3.1.1.6.4 Elasmobranchs and Whales

We believe that there will be no effect to elasmobranchs (giant manta ray, smalltooth sawfish, or scalloped hammerhead shark) or whales (blue, fin, Sei, sperm, and North Atlantic right whale) from agitation dredging. Elasmobranchs and whales are not expected to be injured by low pressure water used in water-injection dredging or a slow moving bed-leveler due to their large size and ability to avoid these dredge equipment types. In addition, whales and scalloped hammerhead shark are not likely to occur in the generally shallower, nearshore areas where agitation dredging will occur.

3.1.2 Entanglement

The presence of flexible materials in the water, such as buoy lines used to mark pipelines or turbidity curtains and in-water lines could create an entanglement risk to mobile species (i.e., sea turtles, fish, elasmobranchs, and whales); however, we believe entanglement from flexible materials in the water associated with activities covered under this Opinion is extremely unlikely to occur. We therefore believe that this route of effect is discountable. The General PDCs in Section 2.2 of Appendix B include specific guidance on the use of in-water lines (e.g., rope, chain, and cable, including the lines to secure the turbidity curtains) and require that all line used will be stiff, taut, and non-looping to minimize the risk of entanglement. If flexible lines are used, they must be enclosed in plastic or rubber sleeves/tubes that add rigidity and prevent the line from looping and tangling. It also requires turbidity curtains and in-water equipment to be placed in a manner that does not entrap species within the construction area or block access for them to navigate around the construction area.

According to the USACE records including those provided on their public dredging website (ODESS), 2 leatherback sea turtles were entangled in flexible nylon rope attached to floating buoy used to mark a pipeline on a project in 2018 near Hilton Head, South Carolina. Both entanglements occurred within 1 week of each other resulting in 1 leatherback sea turtle being released alive and the other being found dead. The USACE contacted NMFS at that time and was instructed to switch all in-water lines to stiff, taut, non-looping in-water lines or flexible lines enclosed in the plastic or rubber sleeves, which they did, and was agreed between the agencies to be an important PDC for this Opinion. We are unaware of reports of any listed species that have been entangled in turbidity curtains or stiff, taut, non-looping in-water lines or flexible lines enclosed in the plastic or rubber sleeves, which supports the belief that the use of these materials reduces the risk of entanglement to make any injury extremely unlikely.

The lines used in relocation trawling also are known to contain flexible, looping line, especially for what are referred to as the lazy lines attached to the relocation trawling nets. However, we believe entanglement in lines other than the net is extremely unlikely and therefore believe that this route of effect is discountable. The relocation trawling PDCs in Section 3.5 of Appendix B state that lazy lines will be designed according to the design specifications in Appendix I , which provide options to make the lazy line taught to minimize the risk of entanglement with captured species. This lazy line guidance was developed to minimize the risk of entanglement to dolphins, which are not addressed in this Opinion, but we believe that these measures will also reduce the risk of entanglement to ESA-listed species. Relocation trawling is closely monitored by a PSO with limited amounts of time that the lines are in the water, as defined by the PSO PDCs in Appendix H and the Relocation trawling PDCs in Section 3.5 of Appendix B. We believe that this further reduces the likelihood of entanglement in lines attached to relocation trawling nets.

Effects to any ESA-listed species that may be entangled in a relocation trawling net as part of a relocation trawling capture are analyzed separately in Section 3.1.1 and Section 6.1.4 of this Opinion.

3.1.3 Capture and Relocation from Relocation and Abundance Trawling

Relocation trawling is method used to minimize the risk of lethal hopper dredging take by sweeping the area around a hopper dredge using a modified shrimp trawl nets to capture and relocate ESA-listed species that may be in the dredging area. While relocation trawling is intended to reduce the occurrence of lethal take from hopper dredging, the process of relocating ESA-listed species is, in and of itself, a form of take under the ESA for those species that are caught. Relocation trawling covered under this Opinion will be monitored by PSOs based on the guidance provided in the PDCs, especially the PSO PDCs in Appendix H that provide handling and reporting guidance for ESA-listed species captured during relocation trawling. Additional PDCs regarding the time and locations where relocation trawling can occur are provided in the General PDCs in Section 3.5 of Appendix B, which limit tow times to 42 minutes to minimize the risk of adverse effects on ESA-listed species, primarily mortality of sea turtles due to forced submergence (National Research Council 1990a) (Epperly et al. 2002). Relocation trawling in the Caribbean is not covered under this Opinion.

Relocation trawling began in the 1980's in Cape Canaveral, Florida. However, relocation trawling has only been used in the action area in limited circumstances. Relocation trawling in the U.S. Caribbean has not previously occurred and is not covered under this Opinion. We reviewed previous projects that used relocation trawling in the action area to evaluate the potential for effects to ESA-listed species. All of the known projects that occurred in the action area that used relocation trawling are provided in Table 13 for reference.

Table 13. Capture Relocation Data within the Action Area

Location	Start Date	End Date	Total Trawling Days	Total Trawling Tows	Atlantic Sturgeon	Green Sea Turtle	Kemp's Ridley Sea Turtles	Leatherback Sea Turtles	Loggerhead Sea Turtle	Total Turtles	Total
Charleston Harbor	3/28/97	5/16/97	49	1,176	0	0	0	0	1	1	1
Morehead City	4/25/97	5/15/97	20	480	0	0	0	0	1	1	1
Myrtle Beach	5/8/97	5/13/97	7	168	0	0	0	0	1	1	1
Canaveral Harbor	10/1/02	10/7/02	7	168	0	14	0	0	55	69	69
Kings Bay	1/24/04	3/18/04	30	544	0	0	0	0	0	0	0
Canaveral Harbor	9/12/04	10/6/04	24	576	0	29	0	0	90	119	119
Broward County Beach Nourishment	5/4/05	5/14/05	10	240	0	0	1	0	24	25	25
Martin County Shore Protection	3/27/05	4/23/05	27	31	0	0	0	0	10	10	10
Ft Pierce Beach Shore Project	4/28/05	6/4/05	37	22	0	0	0	0	3	3	3
Charleston Harbor	12/15/05	1/22/06	38	912	3	0	0	0	0	0	3
Savannah Harbor	3/26/06	4/4/06	9	159	2	0	1	0	0	1	3
Brunswick	12/30/06	1/11/07	12	325	0	0	3	0	3	6	6
Brunswick	3/15/07	3/24/07	9	207	1	0	14	0	17	31	32
Savannah Harbor	1/13/07	2/5/07	53	530	8	0	0	0	0	0	8
Hurricane Ophelia- FEMA sand replacement	3/27/07	3/28/07	1	41	0	0	0	0	0	0	0
Kings Bay Channel	1/11/08	2/23/08	44	1,031	0	0	2	0	1	3	3
Jacksonville Harbor	11/27/07	12/10/07	14	353	0	0	1	0	3	4	4
Brunswick	1/14/08	1/27/08	13	279	10	0	1	0	1	2	12
Brunswick	2/28/08	3/6/08	8	150	0	0	4	0	4	8	8
Brunswick	1/2/09	1/5/09	4	81	0	0	0	0	0	0	0
Brevard County Beach Nourishment	4/1/10	4/17/10	17	419	0	0	0	2	16	18	18
Kings Bay Channel	1/4/10	2/11/10	39	992	29	0	0	0	0	0	29
Brunswick- Bed-leveling study (SER-2013-12117)	3/30/14	4/15/14	20	396	0	0	8	1	8	17	17
Savannah Harbor (SER-2010-05579)	1/7/16	3/31/16	76	1,812	17	0	1	1	0	2	19
Savannah Harbor (SER-2010-05579)	12/2/16	3/31/17	128	3,143	78	2	7	0	10	19	97
Kings Bay	12/4/18	3/31/19	27	305	0	0	2	3	3	8	8

Location	Start Date	End Date	Total Trawling Days	Total Trawling Tows	Atlantic Sturgeon	Green Sea Turtle	Kemp's Ridley Sea Turtles	Leatherback Sea Turtles	Loggerhead Sea Turtle	Total Turtles	Total
Savannah Harbor (SER-2010-05579)	11/30/17	4/1/18	204	5,001	41	2	19	1	30	52	93
Brunswick	1/18/18	3/18/18	59	1,153	79	1	17	1	3	22	101
Savannah Harbor	3/15/18	3/24/18	10	210	0	0	1	1	0	2	2
Dare County (SER-2015-15988)	5/22/17	10/21/17	199	4,599	0	0	2	10	62	74	74
Post 45 Charleston (SER-2014-15433)	1/11/19	4/14/19	92	2,463	12	0	3	4	4	11	23
Bogue Banks (SER-2017-18882)	3/8/19	4/25/19	48	1,493	11	5	3	1	8	17	28
Kings Bay	1/30/19	3/18/19	47	1,128	4	0	0	0	0	0	4
Morehead City	2/28/19	4/11/19	42	1,008	2	0	1	0	0	1	3
All Reports (Fiscal Year 1997-2019)			**1,424**	**31,595**	**297**	**53**	**91**	**25**	**358**	**527**	**824**

A study of the effects of relocation trawling as a mitigation tool to minimize the risk of take from hopper dredging (Dickerson et al. 2008) and data provided by the USACE on relocation trawling take ODESS demonstrate both the risk and benefits of this method. The risks to ESA-listed species of directed take are the stress endured by these species in the process of being trawled and relocated including any potential physical harm during this process and stress that may result in reduced fitness in the form of reduced foraging and reproductive success. Relocation trawling may also have varying levels of effectiveness as a minimization of take with hopper dredging depending on the timing, trawling effort, and project location features. In Section 6.1.4 of this Opinion, we consider these effects to species relocated in the action area.

3.1.3.1 Sea Turtles

Reports show that predominately loggerhead, Kemp's ridley, and green sea turtles are captured during relocation trawling in the Southeast within the action area (listed from highest to lowest reported captures), though there are also limited reports of leatherback sea turtle captures in the action area (ODESS). Therefore, we believe that relocation trawling is likely to adversely affect green (NA and SA DPS), Kemp's ridley, leatherback, and loggerhead sea turtles, which is discussed further in Section 6 of this Opinion.

We believe that there will be no effect to hawksbill sea turtles from relocation trawling. While there have been limited reports of hawksbill sea turtles in relocation trawling (Dickerson et al. 2008) outside of the action area, we do not believe that they will be entrained by hopper dredging covered under this Opinion. Hawksbill sea turtles are closely associated with reef habitat and most prevalent in the action area in South Florida and the U.S. Caribbean. This Opinion does not cover relocation trawling in the U.S. Caribbean. In South Florida, relocation trawling is not covered if coral or coral hardbottom are present to protect ESA-listed corals and Acropora critical habitat (Appendix B, Section ,3.5, Relocate.4) and therefore would also not occur in areas where hawksbill sea turtles occur.

3.1.3.2 Fish

Atlantic sturgeon have been captured in relocation trawling in the South Atlantic portion of the action area, and we expect that shortnose sturgeon, given their life history similarities, may also be captured in relocation trawling as relocation trawling is expanded under the 2020 SARBO (relative to the 1997 SARBO). Therefore, we believe that relocation trawling is likely to adversely affect Atlantic and shortnose sturgeon, which is discussed further in Section 6 of this Opinion.

Nassau grouper may be caught by relocation trawling based on their range and the habitats where they are found within the action area. However, we believe that this route of effect is extremely unlikely and therefore discountable. Since relocation trawling in the U.S. Caribbean is not covered under this Opinion, areas where relocation trawling could occur within the range of this species are limited to South Florida. In South Florida, adult Nassau grouper are associate with reef habitats where relocation trawling will not occur. In nearshore waters where maintenance dredging in navigation channels occurs, Nassau grouper are limited to locations south of (not including) Government Cut in Miami, Florida. We believe that in the Florida Keys, Nassau grouper would be associated with nearshore lagoon habitat with seagrass and mangrove habitat as juveniles and with reef habitat as adults, and therefore would not be in the navigation channel.

3.1.3.3 Elasmobranchs

Smalltooth sawfish and giant manta ray have been reported captured in relocation trawling in areas outside of the action area. While no smalltooth sawfish have been reported captured during relocation trawling in the SARBO action area, 5 smalltooth sawfish were captured in Tampa Bay in 2019 (outside of the SARBO action area). Since relocation trawling in the action area has been limited, these captures indicate that smalltooth sawfish captures are also possible within the SARBO action area, and is discussed further in Section 6 of the Opinion.

We have anecdotal records of giant manta ray captures in relocation trawling associated with dredging in the Gulf of Mexico prior to listing of this species. As relocation trawling in the action area has been limited over the last 15 years, we believe that relocation trawling captures of this species may also occur in the future as a result of the proposed action. Therefore, we believe that relocation trawling is likely to adversely affect smalltooth sawfish and giant manta ray, which is discussed further in Section 6 of this Opinion.

We believe there will be no effect to scalloped hammerhead shark from relocation trawling. This Opinion does not cover relocation trawling in the U.S. Caribbean, which is the only area where the ESA-listed DPS occurs within the action area.

3.1.3.4 ESA-listed Whales

We are unaware of relocation trawling captures of, or other interactions with, ESA-listed whales, from relocation trawling and believe there will be no effect to these species from relocation trawling activities analyzed under this Opinion beyond the potential for vessel strikes (discussed at Section 3.1.4 of this Opinion), or entanglement with other loose lines in the water (discussed in at Section 3.1.2).

3.1.4 Vessel Strike

Mobile ESA listed species may be struck by vessels transiting or working within the action area, as analyzed below for each species. The previous analysis in Section 3.1.1 of this Opinion considered the risk of interaction with equipment as it is dredging and moving material. This section considers the risk of a vessel strike as the vessels and equipment travel within the action area.

3.1.4.1 Sea Turtles

Sea turtles (green, Kemp's ridley, hawksbill, leatherback, and loggerhead) may be physically injured if struck by transiting vessels working on a project. Sea turtles are susceptible to vessel collisions and propeller strikes because they regularly surface to breathe and may spend a considerable amount of time on or near the surface of the water. However, we believe a sea turtle being struck by a vessel operating for a project covered under this Opinion is extremely unlikely and therefore believe that this route of effect is discountable.

Dredging and placement activities and relocation trawling covered under this Opinion will be done by vessels that are slow moving or generally stationary while working, such as barge-mounted equipment, or hopper dredging vessels that are actively dredging or transporting a load of material to a disposal site. We expect that sea turtles would avoid interactions with these slow moving vessels and equipment. The vessels associated with activities covered under this Opinion that are likely to be moving faster are limited to support vessels like crew boats and survey vessels. All vessel operators and crew are required to monitor for the presence of ESA-listed species and follow guidance on distances to avoid them or shut down operations if they are in close proximity (PSO PDCs Section 1 of Appendix H). Vessels used for these activities are the same vessels used for all dredging and placement projects and, although particular projects may result in localized traffic increases while a project is underway, will not result in an increase in vessel traffic within the overall action area. At this time, we are unaware of any sea turtles identified with a vessel strike injury that have been directly related to activities associated with activities that will now be included in the 2020 SARBO proposed action.

3.1.4.1.1 Sturgeon

Sturgeon (Atlantic and shortnose) are susceptible to vessel strike if a deep draft vessel encounters the animals at the sea floor or if the sturgeon moves up into the water column or is sucked into the propeller. We believe that a sturgeon being struck by a vessel associated with a project covered under this Opinion is extremely unlikely. We therefore believe that this route of effect is discountable. NMFS Greater Atlantic Region has stated that reported strandings in the Northeast seem to be related to the large shipping vessels traversing narrow waterways over areas where sturgeon seem to be aggregating. Large vessels have typically been implicated because of their deep draft relative to smaller vessels, which increases the probability of vessel collision with demersal fishes like sturgeon, even in deep water (Brown and Murphy 2010). Also, Miranda and Kilgore (2013) estimated that the large towboats on the Mississippi River, which have a propeller diameter of 2.5 m, a draft of up to 9 ft., and travel at approximately the same speed as tugboats (less than ten knots), kill a large number of fish by drawing them into the propellers. They indicated that shovelnose sturgeon (*Scaphirhynchus platorynchus*), a small sturgeon (~50-85 cm in length) with a similar life history to shortnose sturgeon, were being killed at a rate of 0.02 individuals per kilometer traveled by the towboats.

Historically, vessel strike strandings in the action area have been rare, which was assumed to be because the channels in the action area are wider than those in NMFS Greater Atlantic Region and sturgeon were able to move out of the way of vessel traffic. However, NMFS Southeast Region began dispersing “Report Sturgeon” signage in North Carolina in July 2018, with a particular focus on the Cape Fear River. Since those signs were deployed, 5 sturgeon strandings, showing evidence of a vessel strikes, were reported from Cape Fear River. The increase in reporting may be due to the placement of signs asking citizens to report that were posted June 2018 and the designation of Atlantic sturgeon critical habitat (82 FR 39160, Publication Date August 17, 2017). Additional reports of sturgeon strandings showing signs of vessel strikes have been reported in sturgeon rivers. There is no directed survey for sturgeon strandings and all records are opportunistically reported by the public or resource managers that happen to find an animal, usually on a beach or river bank. A number of the rivers in the Southeast where sturgeon are present are bounded by areas not easily accessible to the public. Thus, a number of sturgeon strandings/carcasses may go unreported simply because they are not detected. We are working with researchers and action agencies to determine which monitoring regimes provide accurate attribution of sturgeon carcasses/strandings to dredge-related activities.

We believe that vessel traffic associated with projects covered under this Opinion are not likely to result in a vessel strike. The rivers in the Southeast tend to be wider than those in the Northeast where vessel strikes occurred and likely provide more room for sturgeon to escape a strike. Sturgeon in the Southeast also generally appear to aggregate in areas outside of heavily trafficked shipping channels, unlikely areas commonly reporting sturgeon vessel strikes in the Northeast. There is currently no evidence that sturgeon are struck by vessels outside rivers. NMFS continues to review available information on strandings to determine if slow moving vessels in sturgeon rivers could result in a sturgeon vessel strike. Currently, we have no reports of vessel strikes by dredge or transit vessels in the action area, and, as stated above, the activities covered under this Opinion will not increase vessel traffic in the action area. In addition, the continued maintenance of navigation channels may allow these areas to maintain clearance between the river bottom where sturgeon are likely to occur and vessels traveling these channels thereby reducing the risk of vessel strikes from all vessels.

3.1.4.1.2 Nassau Grouper

Vessel traffic/boat strikes will not affect Nassau grouper as they are demersal (bottom-dwelling) fish that associate with habitat types where work will not be occurring under this Opinion such as hardbottom and reef habitat in South Florida and nearshore lagoon habitat with seagrass and mangrove habitat in the U.S. Caribbean. We believe that their association with these habitats where work covered under this Opinion will not occur, coupled with their demersal life history lead to a no effect determination. In addition, the General PDCs in Section 2.2 of Appendix B limit work on or near hardbottom, reef, and seagrass habitats and the Coral PDCs protect hardbottom and reef habitat within the range of ESA-listed corals which overlaps with the range of this species (Coral PDCs in Appendix C).

3.1.4.1.3 Elasmobranchs

While both recreational and commercial vessel traffic have been documented to adversely affect protected species, little information exists on vessel interactions with smalltooth sawfish, and scalloped hammerhead shark. Giant manta rays are a recently-listed species (83 FR 2916, Publication Date January 22, 2018) and information is still being collected on the risk of vessel strikes, as discussed below.

- Based on our review of the best available scientific information, we believe that the a vessel strike with a giant manta ray is extremely unlikely, and therefore we believe that this route of effect is discountable. As discussed in the giant manta ray status of the species in Section 4.1.4 of this Opinion, vessel strikes can injure or kill giant manta rays, decreasing fitness or contributing to non-natural mortality (Couturier et al. 2012; Deakos et al. 2011). Giant manta rays can be frequently observed traveling just below the surface and will often approach or show little fear toward humans or vessels (Coles 1916b), which may also make them vulnerable to vessel strikes (Deakos 2010). However, information about interactions between vessels and giant manta rays is limited. We have at least some reports of vessel strike, including a report of five giant manta rays struck by vessels from 2016 through 2018; individuals had injuries (i.e., fresh or healed dorsal surface propeller scars) consistent with a vessel strike. These interactions were observed by researchers conducting surveys from Boynton Beach to Jupiter, Florida (J. Pate, Florida Manta Project, pers. comm. to M. Miller, NMFS OPR, 2018) and it is unknown where the manta was at the time of the vessel strike. The giant manta ray is frequently observed in nearshore coastal waters and feeding at inlets along the east coast of Florida. As recreational vessel traffic is concentrated in and around inlets and nearshore waters, this overlap exposes the giant manta ray in these locations to an increased likelihood of potential vessel strike injury especially from faster moving recreational vessels. Yet, few instances of confirmed or suspected strandings of giant manta ray are attributed to vessel strike injury. This lack of documented mortalities could also be the result of other factors that influence carcass detection (i.e., wind, currents, scavenging, decomposition etc.); however, giant manta rays appear to be able to be fast and agile enough to avoid most moving vessels, as anecdotally evidenced by videos showing rays avoiding interactions with high speed vessels. Some vessel traffic associated with this Opinion will occur in inlets and passes where this species may be found in higher concentrations when dredging these channels; however, vessels involved with relocation trawling or transiting for work covered under this Opinion will be traveling slowly while working in these areas and giant manta rays are mobile species that appear to be able to be responsive to activity in the area and able to move out of the way of at least slow moving equipment. All other, and faster moving, vessel traffic will occur in areas where giant manta rays are expected to be present in much lower concentrations. Due to the expected low concentration of animals in areas where high speed vessel traffic will occur, very limited reports of vessel interactions, and this species' ability to avoid moving vessel traffic outside of confined spaces, we expect that it is extremely unlikely that vessels outside of nearshore inlets and passes will encounter giant manta rays.

- We believe that a vessel strike to smalltooth sawfish is extremely unlikely and therefore conclude that this route of effect is discountable. Smalltooth sawfish are primarily demersal and rarely would be at risk from moving vessels. As vessels need sufficient water to navigate without encountering the bottom, vessels typically transit shoal areas with marginal clearance cautiously (*i.e.*, slowly). Accordingly, we would expect smalltooth sawfish to move out of the way if a vessel came close, and impacts with these species are not anticipated.
- We believe that the risk of a vessel strike to scalloped hammerhead sharks is so low that we expect no effect. The only area that the scalloped hammerhead shark is listed within the action area is in U.S. Caribbean. While there is anecdotal information indicating vessel strikes on shark species do occur, such as when sharks are basking are cruising near the surface, strikes on these particular shark species are not expected due to their preference for offshore pelagic waters outside of the action area, limited time they spend at or near the surface, that they would likely to be seen if at the surface by their large dorsal fin protruding above the water's surface, and due to the limited work about of work expected to be covered under this Opinion in the U.S. Caribbean.

3.1.4.1.4 ESA-listed Whales

ESA-listed whales (blue, fin, Sei, sperm, and North Atlantic right whale) are known to be susceptible to vessel strike collisions that can lead to death; however, we believe that a vessel strike is extremely unlikely and that this route of effect will be discountable based on the PDCs of this Opinion.

North Atlantic right whales are particularly susceptible to vessel strikes due to their cryptic coloring and the lack of a dorsal fin, which make them hard to spot when at the surface. Unlike other whale species in the area, they spend their lives close to shore, often within less than 30 m of water depth where vessel traffic is most prevalent earning them the title of an "urban whale." To address this risk, NMFS published a rule in 2008 (73 FR 60173, Publication Date October 10, 2008) that established vessel speed restrictions to reduce the likelihood of fatal collisions with right whales. Speed restrictions apply in specific locations, primarily at key port entrances, and in certain times in seasonal management areas. The restrictions apply to vessels 65 ft and greater in length. NMFS also established a Dynamic Management Area program whereby vessels are requested, but not required, to either travel 10 knots or less or route around locations when certain aggregations of right whales are detected outside seasonal management areas (73 FR-60173, Publication Date October 10, 2008). Vessels owned or operated by, or under contract to, Federal agencies are exempt from the mandatory application of this rule as a consideration for national security, navigational and human safety missions. As acknowledged in the Final Rule implementing speed restrictions, Federal agencies have an obligation to consult under Section 7 of the ESA regarding how their activities may affect listed species. As provided for in the PDCs of this Opinion, all vessel operations related to projects authorized under this Opinion will follow all of the requirements set forth in this Opinion.

We have numerous reports of vessels strikes on North Atlantic right whales. In 2009, a vessel operator of a 33-ft cruiser vessel reported striking a right whale off New South Wales, Australia. The Australia Department of Environment and Climate Change's Marine Wildlife Situation Report reported that a dead Southern right whale was found a week later 20 km from the collision site and concluded that "it is likely the incidents are linked"(Wiley 2016). This vessel strike report is the smallest vessel known to have killed a right whale and is the reason that NMFS considers all vessels over 33 ft in length to be at risk of a lethal interaction with a right whale. Other reports have been received of vessels that have resulted in lethal vessel strikes to North Atlantic right whale for vessels between 33-65 ft in length (33 ft being the smallest reported lethal strike and 65 ft being the size in which speed restrictions are required under the rule for vessels traveling in designated areas in the United States (73 FR 60173, Publication Date October 10, 2008).

We also have reports of vessel collisions with North Atlantic right whales from vessels similar in size and design as those used by the USACE for surveying dredging operations under this Opinion. In April 2009, the *R/V Auk* struck a right whale while traveling at approximately 20 kts off the coast of Massachusetts (42°11.2'N, 70°33.7'W) (SBNMS 2009; Wiley 2016). Both the *R/V Auk* and *Florida II* (the survey vessel currently operated by the USACE in the action area) are hydrofoil-assisted catamaran vessels with the *R/V Auk* being a 50-ft vessel, while the Florida II is a 62-ft vessel. At the time of the *R/V Auk* strike, 3 experienced whale watch observers were on the fly bridge, the mate was at port side of bridge, and the Captain was at the helm in the cabin. The mate saw the whale roughly 30 ft ahead of the *R/V Auk* and said "whale," but the Captain could not pull back throttles in time to avoid a strike. Whale watch observers saw the whale when it was about 4 ft in front of the *R/V Auk*. The observers noted a fresh wound of multiple (7-8) lacerations and fresh bleeding on the left ventral fluke. They also noted that the whale exhibited abnormal behavior after being struck: the whale was rolling on its left flank while keeping the right fluke tip above the water and exhibiting stressful behavior.

The *R/V Auk* ship strike incident demonstrates that even with well-trained marine mammal observers and vessel operators, all vessels, even research vessels, have the potential to strike cetaceans. In this particular instance, there were 3 dedicated marine mammal observers, but no indication of the animal's presence prior to the initial sighting within 30 ft of the vessel by the mate. The vessel was traveling at approximately 20 knots, which, while not required for a vessel of its size (50 ft), is well above the 10 knots restrictions that were active at the time within the area for larger vessels (greater than 65 ft), and the restrictions that will apply under the PDCs of this Opinion. Winds were 20-23 knots out of the northeast, and wave heights were approximately 4.3 ft, which is not ideal for spotting marine mammals. This is 1 of only 2 instances of research vessel ship strikes ever been reported over the years of cetacean research under MMPA permits. Neither incident appeared to be lethal (Wiley 2016).

We are aware of 2 reports of a hopper dredge collision with a right whale. One report occurred in South Africa in 1984 involving a Southern right whale and the other report occurred in Brunswick Harbor (within the action area) in 2005, though the report is contested by the USACE. The 1984 report stated that a hopper dredge approaching the river-mouth entrance of East London Harbour, South Africa, had report of the presence of a cow-calf pair at the harbor entrance and had been warned to be on the lookout. As the dredge entered the harbor, a Southern Right whale and calf surfaced directly in front of the ship's bow. The calf took the full brunt of the impact and had the full length of the vessel pass over it before the propeller caught it. After attempts by the mother to support the bleeding calf, it made its way across the river to a small sandy beach where it stranded and died shortly afterwards. Photographs of the dead calf show at least 3 separate curved incisions through the dorsal blubber. The mother whale stayed in the area for several hours, and a large crowd of workers had to 'shout and do everything they could' to stop the cow from beaching herself (Daily Dispatch, 17 October 1984). It is unknown the speed at which this vessel was traveling at the time of the incident.

The information available on the 2005 incident in Brunswick Harbor was gathered by NMFS as described below (C. Slay, Coastwise Consulting, pers. comm. to B. Zoodsma, NMFS Southeast Region, February 24, 2005) (M. Zani, New England Aquarium, pers. comm. B. Zoodsma, NMFS Southeast Region, February 24, 2005). On February 24, 2005[28], observers and crew aboard the bridge of the dredge W450 *RN Weeks* felt a "bump" while the dredge was transiting to the Brunswick Harbor Entrance Channel from the offshore dredged material disposal site. The bump or slight shudder was reportedly similar to what might be expected if the vessel had hit something. The ship was moving at 8 kts, on a magnetic heading of 005, at 31° 03.3'N and 81° 16.6'W when this occurred. The observers on the bridge began scanning the water surface as the mate on watch immediately "pulled back" on the controls. What was believed to be the pectoral flipper of a right whale was seen and initially reported as "300-400 ft astern of the ship"[29]. The pectoral flipper made a waving motion above the surface for approximately 5 seconds before the whale submerged. The dredge had coasted to near full stop, but 3-4 ship lengths had been covered (900-1,200 ft). The observers continued to scan with binoculars for 15 minutes, but nothing could be seen at the surface: no whale, no discoloration of the water and no ripples or disturbance. Observers aboard the *BE Lindholm* (another dredge working in the area) reported seeing a whale's blow in the same

[28] Four right whale aerial surveys were being flown during the 2004/2005 right whale calving season. The 2 closest in proximity to the Brunswick Harbor were not flying at the time due to "fog and required pilot down time" (P. Naessig 2/24/2005 email) and "fog and forecasted rain and thunderstorms" (M. Zani 2/24/2005 email).
[29] The distance was later modified to 500 ft (C. Slay 2005 unpub. report).

general area. The central early warning system (EWS) aerial survey team was dispatched to search for a potentially injured whale. The survey team searched for just over 2 hours - they reported unfavorable sighting conditions with light rain and patchy fog during the first hour. During the second hour, the team detected a mother/calf pair at 31.06°N, -81.21°W and within the vicinity of the collision. No other in-water objects that may have caused the dredge to shudder were reported by the aerial survey team.

We believe that the risk of a vessel strike occurring during a project analyzed under this Opinion is very low,[30] since we are only aware of 2 reported interactions with vessels related to dredging, worldwide with North Atlantic or the closely related South Atlantic right whales despite decades of dredging both within the action area and globally. However, the consequences of potential take of a North Atlantic right whale to the small population of the species is high. While we do not normally discuss the status of a species when evaluating effects to a species if the effects from the action are not likely to adversely affect the species, the risk of vessel strikes and potential outcome of a strike to a North Atlantic right whale is unique due to the critical status of the population of this species. Key factors that affect the status of this species include an already low population size that is declining, a decline in the number of calves born annually with none born during the 2017-2018 calving season, an increasing number of years between calving cycles for reproductive females, and evidence of declined health of the reproductive females of this species. Additionally, the action area for this Opinion also includes the only calving grounds for North Atlantic right whales, meaning that smaller calves may be present. Due to their smaller size, calves are at an increased risk of mortality from vessel strikes.

With the considerations outlined above for North Atlantic right whales, the *North Atlantic Right Whale Conservation Plan* (Appendix F) was developed cooperatively by NMFS, USACE, and BOEM and for a project to operate under the 2020 SARBO, the conservation measures outlined in this plan must be followed. Specifically, the Plan provides funding for aerial surveys with the portions of the action area where North Atlantic right whales may be present, which includes an expansion of current aerial survey coverage area. The SARBO 1997 survey area was limited to portions of northeast Florida and Georgia during times when this species may be present. The expanded area will extend north of the SARBO 1997 survey area to cover the remaining portions of the action area where this species may be present. The *North Atlantic Right Whale Conservation Plan* in Appendix F describes how, when, and where surveys will be completed by continuing the existing aerial survey and adding up to 2 additional survey teams focused on detecting North Atlantic right whales migrate from their northern feeding areas outside of the action area to the southern calving area in the action area. The *North Atlantic Right Whale Conservation Plan* also details how notification of the presence of these whales will be communicated. Reports of North Atlantic right whale presence from both the aerial surveys and reports from crew working on projects covered under this Opinion if this species is observed will then be broadcast to other commercial mariners in the area triggering speed restrictions for vessels associated with the projects covered under this Opinion within a specified distance of the whale sighting and will be sent as an alert to commercial vessels in the area to be on the lookout for their presence. Increasing the surveyed area also increases the probability of observing all whales and

[30] For discussion about some of the background of the approach to North Atlantic right whales, see Section 1.2.5 of this appendix.

calves that enter the southeast, which adds increased confidence in total population numbers and calves born each year.

The *North Atlantic Right Whale Conservation Plan* (Appendix F) also requires that a PSO trained to identify ESA-listed species be aboard all hopper dredges to observe for ESA-listed species and alert the captain of their presence to minimize the risk of a vessel strike. If a North Atlantic right whale is identified, whether by shipboard observation or aerial survey, all vessels within 38 nautical miles (nmi) and over 33 ft in length that are associated with a project covered under this Opinion will be required to slow to 10 knots, as required in the North Atlantic Right Whale Plan (Appendix F) when working when and where North Atlantic right whales may be present in the action area. The distance of 38 nmi was chosen based on the distance that the vast majority of North Atlantic right whales travel in 24 hours. To determine the appropriate distance, 177 pairs of North Atlantic right whale sightings in the Southeast calving area that were 24 hours apart were reviewed. The average distance between sightings 24 hours apart was 14.56 nmi (standard deviation of 12.32 nmi), 95th percentile was 38 nmi, and the maximum recorded distance was 85 nmi in 24 hours (Gowan 2014 unpublished analysis). The 10 knot speed restriction is based on the information presented in NMFS's *Final Rule to Implement Speed Restrictions to Reduce the Threat of Ship collisions with North Atlantic Right Whales* (73 FR 60173, Publication Date October 10, 2008) that determined that 10 knots was the appropriate speed to reduce mortality resulting from a ship strike. The rule also noted that the strike risk posed by a conventional ship moving 20-25 knots could be reduced by 30% by slowing to 12 or 13 knots, and by 40% at 10 knots, due to the whales' increased ability to detect and avoid approaching vessels.

Because there are so few North Atlantic right whales, and much of the vessel traffic associated with the proposed action will take place outside of areas and times when North Atlantic right whales may be present, the likelihood of collisions is already very rare. We believe that the implementation of these additional protective measures in the PDCs further reduces the possibility of a vessel strike. When the rarity of occurrence is combined with the requirements of the *North Atlantic Right Whale Conservation Plan*, we believe a vessel strike is extremely unlikely to occur.

We believe that a vessel strike to the other ESA-listed whales in the action area (blue, fin, Sei, and sperm) is also extremely unlikely to occur. Therefore, we believe that this route of effect is discountable. These whales tend to be deeper water species typically found off the continental shelf in waters deeper than where work covered under this Opinion will generally occur, except placement in ODMDS and borrow area dredging. Additionally, while the PDCs in the *North Atlantic Right Whale Conservation Plan* are specifically designed for the protection of that species, the PDC requirement for slower speed vessels and observers on dredging vessels provide protection to all whale species, if present, by improving awareness of the potential presence of North Atlantic right whale in the area by aerial surveys and imposing speed restrictions when and where they may be present.

3.1.5 Species Interaction with the Placement of Material

This Opinion covers placement of material by multiple types of equipment, including side-cast, split/hull hopper placement, and pipeline placement described in Section 2 of this Opinion. Generally, all of these methods are used to deposit material through the water column to the sea floor or to place it on land for upland disposal or beach nourishment. Placement may occur in a number of areas discussed in Section 2.4 of this Opinion, including sand placement on beaches, nearshore placement, beneficial use placement, ODMDS, and upland placement. The potential for interaction from these types of equipment while they are depositing the material is limited to the potential of the species being directly below the material as it is passing through the water column and landing on the sea floor.

We believe that risk of a mobile species being caught in the discharge through the water column and buried on the sea floor is so low as to make the route of effect discountable. These mobile species would be able to detect the presence of the material being deposited and avoid being harmed by its placement. Placement in an open ocean environment such as an ODMDS would allow room for species to move away from and around the placement. In addition, the general PDCs require that crew members be aware of the species that could occur in the work area and to monitor for their presence (General PDCs Section 2.1 and the PSO PDCs in Appendix H). If ESA-listed species are spotted within the distances provided in the PDCs for the different species in the action area, activities may not resume until the protected species has departed the project area of its own volition (PSO PDCs Section 1 in Appendix H).

3.1.6 Blocked Access by Construction or Material Placement

Mobile ESA-listed species may be temporarily unable to use a project site for forage and shelter habitat due to avoidance of dredging activities, related noise, and physical exclusion from areas; however, we believe that species will avoid these areas, and any effects would be temporary and insignificant for Nassau grouper, elasmobranchs and whales. All activities covered under this Opinion are limited to confined areas where similar type of habitat is nearby which would support the same activities. Thus, any animals disrupted by the activities covered under this Opinion would be expected to continue to conduct the same activities in the surrounding areas not disrupted by activities covered under this Opinion. Species may also be deterred from entering an area by increased noise, which is discussed separately in Section 3.1.8 of this Opinion.

Activities conducted under this Opinion could affect movement and access to habitat of these mobile species; however, we believe the effect will be insignificant because the General PDCs (General PDC Section 2.2, Appendix B) require that all work, including equipment, staging areas, and placement of materials, will be done in a manner that does not block access of ESA-listed species from moving around or past construction.

Of the mobile species considered in this Opinion, only Nassau grouper, sea turtles, and sturgeon use nearshore environments for important life cycle functions. However, material placement will not impact nearshore Nassau grouper habitat, as the General PDCs in Section 2.2 of Appendix B limit work on or near hardbottom, reef, and seagrass habitats. The potential for impacts to nesting turtles and sturgeon, from blocked access are discussed specifically, below.

3.1.6.1 Sea Turtles

Loggerhead, NA DPS green, and leatherback sea turtles nest on Florida beaches and hawksbill, SA DPS green, and leatherback sea turtles nest on beaches in the U.S. Caribbean. Female sea turtles migrate to nesting beaches to lay eggs and hatchlings migrate away from these beaches. The placement of materials or presence of equipment in front of (i.e., waterward of) nesting beaches could interfere with or obstruct sea turtles' ability to access or leave the beach; however, we believe that there will be an insignificant effect to sea turtles from placement activities covered under this Opinion restricting access to or from sea turtle nesting beaches. This Opinion includes PDCs designed to protect sea turtles' access to nesting beaches by ensuring that materials are not piled high in nearshore environments, such that these materials would block sea turtle access to and from nesting beaches (General PDC Section 2.2, Appendix B). In addition, beach nourishment projects are not covered under this Opinion in the U.S. Caribbean.

Some sea turtles may also potentially be subject to disorientation from equipment lighting near sea turtle nesting beaches; however, we believe that any effects would be insignificant. Female sea turtles approaching the beach to nest could be deterred from nesting by bright lights in the nearshore environment. Hatchlings emerging from their nests could be attracted away from the shortest path to the water and instead crawl or swim toward the bright lights of a nearshore hopper dredge or anchored pumpout barge (instead of crawling or swimming seaward toward the open horizon), thus increasing their exposure time to predation. The General PDCs in Section 2.2 of Appendix B state that all lighting near sea turtle nesting beaches will be shielded and minimized to the maximum extent possible consistent with vessel personnel safety and U.S. Coast Guard navigation requirements, to reduce potential disorientation effects, potential reduced or aborted nesting, and potential increased hatchling mortality from increased exposure to predators. In addition, activities near nesting beaches will occur in rare instances, and may require additional coordination with the USFWS.

Outside of nesting beaches, sea turtles may be temporarily unable to use a project sites for forage and shelter habitat due to avoidance of dredging activities, related noise, and physical exclusion from areas; however, as for other mobile species, we believe that any effects would be temporary and insignificant for the reasons stated in 3.1.6.

3.1.6.2 Sturgeon

Activities covered under this Opinion occur in rivers where sturgeon (Atlantic and shortnose) are migrating into and through on their way to spawning habitats or other areas required to fulfill life history functions. To protect these migratory pathways, the Sturgeon PDCs require: no in-water placement of dredge material in sturgeon rivers, including side-cast dredging and nearshore placement. The PDCs also require that all dredging operations, including related equipment, and projects conducted by other entities in the vicinity, do not block more than 50% of the sturgeon river width to allow safe passage of sturgeon. While work within the sturgeon spawning river will occur when sturgeon are present, we believe that limiting work to ensure that at least half of the river width remains free of work will mean that sturgeon will still be able to migrate unimpeded during the duration of construction. Sturgeon will be able to move unimpeded following the completion of any dredging operations. As discussed in Section 3.1.1 of this Opinion, sturgeon appear to be undeterred by the presence of dredging equipment and that activities covered under this Opinion would have an insignificant effect on their ability to move through the rivers.

Outside of sturgeon rivers, sturgeon may be temporarily unable to use a project sites for forage and shelter habitat due to avoidance of dredging activities, related noise, and physical exclusion from areas; however, as for other mobile species, we believe that any effects would be temporary and insignificant for the reasons stated in 3.1.6.

3.1.7 Habitat alteration from Activities Covered under this Opinion

Dredging or placing material alters existing habitat within the project footprint, which may affect ESA-listed species that use that habitat. Based on the activities covered and the PDCs that limit them, we believe habitat alteration is not likely to adversely affect or will have no effect on ESA-listed species, as discussed by species below (however, muck dredging is addressed first, for all species, as we expect no effects to listed species from this activity). The PDCs were designed to minimize the effects from activities covered under this Opinion, including minimizing the loss of foraging or refuge habitat used by ESA-listed species and to minimize the loss of areas that non-mobile species can recruit, such as hardbottom used by ESA-listed corals and sand areas used by Johnson's seagrass. In addition, the array of individual projects covered under this Opinion are expected to be separated both temporally (over the life of this Opinion) and spatially (throughout the entire action area), such that we do anticipate overlap of individual projects happening at the same time in the same place. Habitat alteration resulting from projects covered under this Opinion are expected to be confined to the dredge or placement area or areas where materials and equipment are placed during construction. Species using these areas will still be able to forage or seek refuge in nearby areas outside of active project sites.

3.1.7.1 Muck Dredging/Restoration Dredging

The areas to be muck dredged under this Opinion are areas covered in accumulated organic material and thus would not support foraging or refuge habitat used by ESA-listed species, due to the poor sediment and water quality in these areas. Therefore, we believe muck dredging will have no effect on ESA-listed species by affecting their forage resources. Muck dredging may actually make the habitat in this area more suitable for bottom and near bottom mobile species, including sea turtles, sturgeon, and elasmobranchs, by allowing foraging resources to recruit into these areas in the future. Additionally, if the area has poor water quality as a result of accumulated sediments, ESA-listed bottom and near bottom mobile species are unlikely to be present. Whales do not forage on bottom dwelling resources and would not be effected by muck dredging changing foraging resources. Johnson's seagrass and ESA-listed corals are non-mobile species that do not need foraging or refuge habitat, but do use new areas for recruitment. These non-mobile species would not be found in muck environments or recruit to them. However, Johnson's seagrass may benefit from the removal of muck if it restored an area to sand that could be used for recruitment into new areas.

3.1.7.1.1 Sea Turtles

We believe that habitat alteration resulting from activities covered under this Opinion is not likely to adversely affect sea turtles (green, Kemp's ridley, hawksbill, leatherback, and loggerhead) for the reasons provided below.

Seagrass impacts
Seagrasses are an important foraging resource for green sea turtles and may be removed by maintenance dredging in shallow waters either by directly removing seagrasses within the maintenance dredge footprint, burial by placement of dredged material, or burial of seagrasses from turbidity and sedimentation generated by nearby dredging or material placement. We believe the loss of seagrass will have an insignificant effect on green sea turtles, since dredging and

placement activities are limited to activities that are either a continued use of previously dredged areas or material placement locations where seagrasses have previously been removed, or limited to new areas that will avoid seagrasses to the maximum extent practicable, as described by the PDCs (General PDCs in Section 2.2 of Appendix B). In addition, all work that may generate turbidity will be completed in a way that minimizes the risk of turbidity and sedimentation reaching non-ESA-listed corals, sponges, and other natural resources by limiting turbidity generated through PDCs in this Opinion. For example, the Coral PDCs in Appendix C limit the type of equipment used and the length of time dredging will occur when coral hardbottom is present to prevent sedimentation burial of nearby coral and coral hardbottom, which is expected to also protect non-listed corals and sponges that also use this habitat. In addition and adherence to regional water quality standards outside NMFS jurisdiction that regulate turbidity limit the amount of sedimentation effects outside of the project footprint (General PDCs in Section 2.2 of Appendix B). If any impacts do occur to sea turtle foraging resources in the area, we expect that similar habitat will be available in the surrounding areas and that the impact will have an insignificant effect on the ability for sea turtles to forage.

Limestone outcroppings and worm-rock reefs
These hardbottom resources are important developmental habitat for juvenile green turtles. We believe there will be no effect to this habitat, and therefore no effect to the species, because the General PDCs in Section 2.2 of Appendix B state that areas not previously dredged or areas where in-water material placement has not previously occurred will avoid the removal or placement of materials on nearshore or surf-zone, low-profile hardbottom outcroppings (e.g., worm-rock reef [sabellariid worm reefs] and eolianite, granodiorite).

Other hard structure foraging and refuge habitat used by sea turtles
All sea turtles use other hardbottom besides limestone outcropping and wormrock for foraging and refuge. We believe these activities covered under this Opinion will have an insignificant effect on a sea turtle's ability to forage or seek refuge in this habitat. Green sea turtles forage on algae and hawksbill sea turtles forage on sponges growing on hard structures including hardbottom, natural relief areas and man-made structures such groins and breakwaters that may be located during activities covered under this Opinion. The General PDCs in Section 2.2 of Appendix B state that all new dredging and placement will avoid areas with significant non-coral hardbottom, defined as an area with a horizontal distance of 150 ft that has an average elevation above the sand of 1.5 ft or greater and has algae growing on it. These areas will be identified prior to commencing work to ensure that all sand removal projects provide a 400 ft buffer from all equipment, to minimize any risk of impacts from sedimentation to hardbottom. In addition, the PDCs within the range of ESA-listed corals protect coral hardbottom and reef habitat from being removed or buried by sedimentation generated by nearby dredging or material placement (Coral PDCs Section 2 in Appendix C), which would also protect this habitat for sea turtles. These PDCs serve to protect both the hard structure resource and ensure work is occurring a sufficient distance from natural hardbottom.

This Opinion does not cover the installation or removal of man-made structures such as groins or breakwaters that may support algae or sponges used by sea turtles for foraging and any turbidity effects to these resources in the surrounding area is expected to be minimal and temporary since these are vertical surfaces that would not be buried by sedimentation. For example, juvenile green sea turtles are known to forage on macroalgae growing on riprap in Canaveral Harbor (Redfoot and

Ehrhart 2013). According to the study, 99.4% of the turtles captured in the Trident Turning Basin in Canaveral Harbor from 1993-2011 were juvenile greens. The authors report that Trident Basin appears to be at or near its carrying capacity for green sea turtles. We expect that other areas similar to Canaveral Harbor that have extensive riprap at the entrance to a channel or harbor also support large numbers of juvenile green sea turtles if they also have a large amount of available algae. Because activities covered under this Opinion will not involve removal or other alteration of these structures, we believe that the proposed action will have an insignificant effect on sea turtles' ability to use these foraging resources.

Other sea turtle foraging
Dredging and placement activities may remove or bury areas inhabited by sea turtle prey species, including the crustaceans and mollusks that serve as prey for loggerhead sea turtles and the fish, jellyfish, shrimp, and mollusks that serve as prey for Kemp's ridley sea turtles. These effects are limited in area, temporary, and benthic foraging resources are expected to recolonize these areas. The recovery time of an area varies by the size of the impact, water temperature, and sediment type and can range from 6-8 months in estuarine areas with mud to 2-3 years in areas with sand and local disturbances by waves and currents (Newell et al. 1998). Areas frequently maintained likely lack these resources and may not have ample time between maintenance cycles to recover. However, other areas such as where dredging or material placement occur less frequently are likely to recover and swimming prey such as jellyfish as well as mobile prey like shrimp, may recover more quickly moving from surrounding undisturbed areas. We believe that sea turtles can continue to forage in surrounding areas until the dredge or placement location recolonizes, therefore we believe the effect of any temporary loss of these foraging resources will be insignificant.

Sea turtles in channel
Sea turtles may also use channels to thermal regulate by entering deeper channels in the summer to avoid warmer surface waters and entering deeper water in the winter where waters may be warmer than winter surface temperatures. The inability to access these channels during dredging would be temporary and dredging would not occur throughout the entire reach of a channel at the same time, allowing other areas of the channel to remain available to sea turtles to thermal regulate. Therefore, the inability to access a portion of the channel during dredging will have an insignificant effect on sea turtle's ability to use the area to thermal regulate.

3.1.7.1.2 Sturgeon

We believe that habitat alteration resulting from activities covered under this Opinion is not likely to adversely affect sturgeon (Atlantic and shortnose) for the reasons provided below.

Atlantic and shortnose sturgeon use the rivers identified in the Sturgeon PDCs in Appendix E, estuarine areas (e.g., bays and harbors), and open-water marine environments. Historically, it was assumed that shortnose sturgeon tended not to leave riverine waters (i.e., venture beyond the freshwater-saltwater interface); however, in a recent report by the South Carolina Division of National Resources and Georgia Division of National Resources, shortnose sturgeon were detected as far as 12.4 miles from the mouths of their spawning rivers in those states (Post et al. 2017a). While we believe that it is still less common to find shortnose sturgeon in open-water marine environments, we consider the potential effects to both shortnose and Atlantic sturgeon in these areas.

Sturgeon in Rivers
The Surgeon PDCs in Section 2 in Appendix E limit activities covered under this Opinion to portions of the river where spawning does not occur; therefore, we believe there will be no effect to sturgeon spawning habitat. Table 56 provides a list of the upstream limit of work covered under this Opinion. Activities may occur above this upstream limit on a project-by-project basis under the Supersede procedures outlined in Section 2.9.5 of this Opinion. However, sufficient evidence must be available that the project will not alter spawning habitat or affect spawning sturgeon, eggs, or other early life stages of sturgeon. If new information becomes available regarding sturgeon spawning locations, Table 56 will be updated to continue to protect spawning habitat, as described in the Sturgeon PDCs in Appendix E and PDC modification in Section 2.9.5.3 of this Opinion.

As sturgeon hatch, grow, and move downstream toward estuarine habitats, it is essential for developing fish to forage in areas of soft substrate and seek shelter along the way as they are slowly exposed to changes in salinity to allow physiological adaptations to higher salinity waters. Thus, this feature is necessary for juvenile foraging and physiological development.

We anticipate that habitat alteration resulting from the activities covered under this Opinion will have an insignificant effect on a sturgeon's ability to forage or seek refuge in spawning rivers. Dredging projects considered under this Opinion will remove substrate that contains sturgeon prey. Dredging has been shown to result in a significant drop in species diversity, abundance, density, and biomass within the dredge footprint (Newell et al. 1998). How quickly those areas recover is dependent upon which benthic community was there initially, the surrounding soil type, project latitude, whether the area was regularly disturbed previously, and whether the benthic community is already adapted to frequent disturbance (Newell et al. 1998). Generally, recolonization rates are faster for fine-grain sediments (e.g., mud and clay) and longer for larger grain sediments (e.g., sand and gravel) (Newell et al. 1998). The recovery time can range from 6-8 months in riverine areas with mud. Riverine areas used by sturgeon in the action area typically have fine muds, clays, small grain sands, which are likely to be colonized by large populations of opportunistic species that are capable of rapid colonization within months of space being made available for colonization and growth (Newell et al. 1998). These individuals tend to have selected for maximum rate of population increase, high fecundity, dense settlement, rapid growth and relatively short life spans (e.g., polychaete worms and amphipods), making them well suited to rapid colonization of environments where space has been left by a previous catastrophic mortality, whether caused naturally or by man (Newell et al. 1998). Many of these first colonizing species are prey items for sturgeon.

These riverine habitats are subject to regular disturbance under natural conditions prior to dredging. Recovery of the "normal" community in disturbed deposits such as muds can be achieved within months of cessation of dredging or disposal of spoils (Bolam and Rees 2003; Newell et al. 1998; Wilber and Clarke 2007). This recolonization response is supported by studies reported in Wilber and Clarke (2007). The authors report the recovery time for a number of dredged channel sites around the world and in different climates (i.e., sub-tropical, temperate, cold). The 3 dredged channel sites located in the Southeast United States showed recovery times of 3-6 months (Stickney and Perlmutter 1975; van Dolah et al. 1984; van Dolah et al. 1979; Wilber and Clarke 2007).

Dredging in the navigation channel occurs on an as needed basis with some areas occurring more frequently and some only occurring every few years. Maintenance dredging in other areas of the river such as ports, berths, marinas, and around other existing structures will occur as needed. Dredging anticipated to be completed in the next 5 years provided by the USACE, demonstrates that dredging interval of all projects will generally be longer than 6 months even in frequently dredged areas. In other words, even frequently dredged areas will not be disturbed again by dredging for at least 6 months. This should allow sufficient time for these areas to be predominately or fully recolonized before the next dredge event. While the areas adjacent to dredging will not be removed, they may receive sedimentation from the turbidity generated during dredging, especially from agitation dredging such as water-injection dredging. There is evidence that benthic communities in areas immediately adjacent to dredging may benefit from the release of organic materials caused by dredging (Newell et al. 1998). Newell et al. (1998) report several studies documented enhanced species diversity and population density of organisms in areas immediately adjacent to dredge sites due to these releases (Biggs 1968; Ingle 1952; Jones and Candy 1981; Oviatt et al. 1981; Poiner and Kennedy 1984; Sherk Jr. 1972; Stephenson et al. 1978; Walker and O'Donnell 1981). Because we anticipate dredged areas will have an opportunity to recover, and areas immediately adjacent dredging areas will not be removed (and possibly enhanced), we believe riverine foraging resources used by sturgeon will continue be able to fulfill its ecological role/function.

Placement of dredge material in sturgeon rivers is not covered under this Opinion (e.g., nearshore placement and side-cast dredging) and therefore would not result in direct burial of prey resources or fill deep holes that may be used by sturgeon for refuge.

Riverine areas that include hard substrate and rock occur further upstream of the areas where activities covered under this Opinion will occur based on the upstream limits in the Sturgeon PDCs in Table 56 in Appendix E.

Sturgeon in Estuaries and Marine Environment

We believe the activities covered under this Opinion will have an insignificant effect on a sturgeon's ability to forage. Sturgeon are opportunistic feeders. Unlike rivers where foraging habitat is relatively confined and discrete, when sturgeon are in larger estuarine and marine systems they are able to forage over large areas. We anticipate they will be able to locate prey beyond the immediate areas where work will occur. Sturgeon are not expected to forage in areas where beach nourishment or nearshore placement associated with beach nourishment occur due to the shallow depths and high energy areas associated with coastal beaches. It is possible they may forage in borrow sites, ODMD placement sites, or other nearshore placement location. Dredging or dredge spoil placement in these areas effects foraging resources availability, but this is expected to be temporary since benthic invertebrate populations in dredged areas are expected to recover. As noted previously, the recovery time of an area varies by the size of the impact, water temperature, and sediment type. Recovery times can range from 6-8 months in sturgeon estuarine areas with mud to 2-3 years in areas with sand and local disturbances by waves and currents (Newell 1998) such as those typically found in borrow areas, near beaches, and in entrance channels. Areas outside of the sturgeon rivers may include both mud (discussed above in the sturgeon riverine analysis) and sand environments. In frequently maintained areas like sandy entrance channels to ports and harbors, these channels likely lack sturgeon prey resources and may not have ample time between maintenance cycles to recover since sandy areas are slower to recover. However, other areas such as where dredging or material placement occur less frequently, are likely to recover within a few years. Since sturgeon are opportunistic foragers, they will continue to be able to use surrounding areas as these areas recover. Also, these areas are not dredged or filled simultaneously and typically larger areas like an offshore borrow site or ODMDS only uses a portion of the available area within the borrow or ODMD site for each project, leaving the remaining areas available for foraging. Beneficial use placement that includes thin-layer placement in potentially large areas was not included in this Opinion, and therefore must be evaluated in a separate individual Section 7 consultation or under the Supersede procedures outlined in Section 2.9.5 of this Opinion to ensure that large areas outside of sturgeon rivers would not be covered thereby removing potential foraging areas needed by sturgeon either staging for a spawning run or returning from spawning.

3.1.7.1.3 Nassau grouper

We believe the activities covered under this Opinion will have an insignificant effect on a Nassau grouper's ability to use habitat for foraging and seeking refuge. Nassau grouper are associated with habitat types where work will not be occurring under this Opinion, such as hardbottom and reef habitat in South Florida and nearshore lagoon habitat with seagrass and mangrove habitat in the U.S. Caribbean. The General PDCs in Section 2.2 of Appendix B limit work on or near coral and seagrass habitat and the Coral PDCs protect hardbottom and reef habitat within the range of ESA-listed corals which overlaps with the range of this species (Coral PDCs in Appendix C). Even if maintenance dredging temporarily removed some non-listed seagrasses, all work would be contained to the maintained navigation channels and the surrounding habitat would continue to support foraging and refuge habitat for this species.

3.1.7.1.4 Elasmobranchs

Giant Manta Ray

We believe that the activities covered under this Opinion will have no effect on giant manta ray's ability to use foraging or refuge habitat. It is a migratory species, and seasonal visitor along productive coastlines with regular upwelling, in oceanic island groups, and near offshore pinnacles and seamounts (83 FR 2916, Publication Date January 22, 2018). The timing of these visits varies by region and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior. Although the giant manta ray tends to be solitary, they aggregate at cleaning sites and to feed and mate. They are commonly encountered on shallow reefs while being cleaned or are sighted feeding at the surface inshore and offshore. They are also occasionally observed in sandy bottom areas and seagrass beds. The species has also been observed in estuarine waters near oceanic inlets, with use of these waters as potential nursery grounds. Giant manta rays appear to exhibit a high degree of plasticity in terms of their use of depths within their habitat. During feeding, giant manta rays may be found aggregating in shallow waters at depths less than 10 m. However, tagging studies have also shown that the species conducts dives of up to 200 -450 m and is capable of diving to depths exceeding 1,000 m.[31] None of the activities covered under this Opinion will alter the availability of zooplankton in the water column used by this species for foraging. Some of the activities such as channel maintenance dredging and beach nourishment will occur in areas used by this species; however, they are not expected to alter the habitat in a way that would affect this migratory species

[31] https://www.fisheries.noaa.gov/species/giant-manta-ray

that uses a wide range of habitat types described above. If this species were present near an active project site, we assume that this mobile species would move and seek out similar available habitat nearby.

Smalltooth sawfish

We believe the activities covered under this Opinion will have an insignificant effect on a smalltooth sawfish's ability to use habitat to forage and seek refuge. For the first several years of their lives, juvenile sawfish mature, they move to deeper water areas where dredging or material placement could alter foraging habitat. The activities covered under this Opinion within the range of smalltooth sawfish are limited to generally areas already dredged or where material has previously been placed, leaving the surrounding areas available for foraging until areas that are disturbed can recover or recolonize with foraging resources. We believe that smalltooth sawfish can continue to forage in surrounding areas until the dredge or placement location recolonizes, therefore we believe the effect of any temporary loss of these foraging resources will be insignificant. Additionally, smalltooth sawfish would still be able to forage or seek refuge in nearby areas outside of active project sites.

Scalloped hammerhead shark

Within the action area, only the DPS of scalloped hammerhead sharks occurring in the U.S. Caribbean is ESA-listed. Scalloped hammerhead shark are highly mobile and partly migratory, and are likely the most abundant of the hammerhead species (Maguire et al. 2006). Both juvenile and adult scalloped hammerhead sharks occur as solitary individuals, pairs, or in schools (Compagno 1984). Adult aggregations are most common offshore over seamounts and near islands (Bessudo et al. 2011; CITES 2010; Compagno 1984; Hearn et al. 2010), which are outside of the action area of this Opinion. Neonate and juvenile aggregations are more common in nearshore shallow water nursery habitats (Bejarano-Álvarez et al. 2011; Diemer et al. 2011; Duncan and Holland). This species of shark is a high trophic level predator (Cortés 1999) and an opportunistic feeder with a diet that includes a wide variety of mobile species such as bony fish, octopi, cuttlefish, squid, crabs, lobsters, and rays (Bush 2003; Compagno 1984; Júnior et al. 2009; Noriega et al. 2011). In the listing rule, we identified commercial fishing as one of the primary threats to the Central and SW Atlantic DPS of scalloped hammerhead shark, and found it was unlikely that loss of habitat was contributing to the species' extinction risk (79 FR 38213, Publication Date July 3, 2014). We believe that the limited and temporary habitat alteration covered under this Opinion from maintenance and borrow area dredging and material placement (e.g., ODMDS) will have an insignificant effect on the species, since most work occurring in the U.S. Caribbean where this species is listed will be limited by restrictions of work occurring near ESA-listed corals in that area, where scalloped hammerhead sharks are rarely present due to their preference for offshore pelagic waters outside of the action area. Even if present, we assume that this mobile species would move and seek out similar available habitat nearby, outside of an active project site.

3.1.7.1.5 Whales

We believe there will be no effect to whales (blue, fin, sei, sperm, and North Atlantic right whale) from habitat alteration resulting from activities covered under this Opinion. Whales are pelagic species that do not forage on benthic resources and therefore will not be affected by changes in sediment from dredging or material placement. North Atlantic right whales use the action area for calving and move north to forage. While in the action area, North Atlantic right whale mothers do not forage while nursing their calves. The activities covered under this Opinion will not alter the depth of waters used by whales, as dredging will only maintain or increase depth in an area and in-water material placement will only result in a minor decrease in the overall depth in an area that is expected to be undetectable to whales using this habitat. If whales are present near an active project site, we assume that this mobile species would move and seek out similar available habitat nearby. Additionally, the PDCs require that activities cease if a whale is observed; therefore, we would not expect that the proposed action will interfere with ESA-listed whales use of the action area, including North Atlantic right whale calving. In addition, blue, fin, sei, and sperm whales occur in deeper water environments away from most activities covered under except placement in ODMDS and borrow area dredging.

3.1.8 Sound Generated by Projects Covered under this Opinion

Activities under this Opinion will generate both in-water and above-water sounds with the potential to effect ESA-listed species considered in this Opinion. To determine the level of effects to ESA-listed species, we need to know the level of sound generated, if that sound can be detected by a particular species, and the acoustic threshold at which the species responds to the sound. Sound can effect animals either by the sound waves causing physical injury to the animal or by altering their behavior if the sound source is within their hearing range. First we discuss the hearing range of ESA-listed species that may occur in the action area (Section 3.1.8.1 of this Opinion), followed by the thresholds in which acoustic effects may occur (Section 3.1.8.2 of this Opinion), and then the sound levels expected from the activities and how we believe these species will respond to it (Sections 0-Section 3.1.8.5 of this Opinion). Our consideration of how species respond to a sound source is divided by the type of sound generated by activities covered under this Opinion including (1) the operation of equipment discussed in Section 2.5, including geotechnical surveys discussed in Section 2.6 of this Opinion; (2) geophysical surveys discussed in Section 2.6 of this Opinion; or (3) aerial surveys conducted using planes to detect North Atlantic right whales (Appendix F) discussed in Section 2.7.3 of this Opinion.

3.1.8.1 Hearing Range of ESA-listed Species in the Action Area

If a sound generated is within an animal's hearing range, it may have varied behavioral responses to the sound, depending on the level of sound, how the animal is using the area at the time of sound exposure, site specific conditions, and life stage of the animal at the time of exposure. Different species have different life histories and sound-detection capabilities that could influence their response, in addition to a different set of potential behavioral responses that may be triggered by a given stimulus (e.g., some may try to hide in the area while others may swim away). Although the behavioral response of avoidance to sound is advantageous for preventing potential injury from the exposure to the sound, avoidance behavior may disrupt or interfere with feeding, mating, migration, or sheltering, or may increase other risk to individuals (e.g., via predation). Not all individuals are likely to have an avoidance response. Despite exposure to sounds that may cause others to move away and abandon the area altogether, some individuals may be biologically motivated to remain in a habitat for feeding, sheltering, mating, or other biologically important reasons or because they are using the area as an established pathway between habitats.

The hearing ranges of ESA-listed species that may occur in the action area are provided in Table 14 below, followed by a discussion of how we determined the hearing range for species with limited available information on their specific hearing range.

Table 14. Hearing Ranges of ESA-Listed Species

Species or Group	Hearing Range
Sea turtles (green, Kemp's ridley, hawksbill, leatherback, loggerhead)	30 Hertz (Hz) to 2 kHz
Elasmobranchs (giant manta ray, scalloped hammerhead shark, smalltooth sawfish)	20 Hz to 2 kHz
Fish (Atlantic and shortnose sturgeon, Nassau grouper)	100 Hz to 2 kHz
Low-frequency cetaceans (blue, fin, and sei)	7 Hz to 35 kHz
North Atlantic right whale	10 Hz to 22 kHz
Mid-frequency cetaceans (sperm whale)	150 Hz to 160 kHz

3.1.8.1.1 Sea Turtles

Sea turtles are low frequency hearing specialists, typically hearing frequencies from 30 Hz to 2 kHz, with a range of maximum sensitivity between 100 to 800 Hz (Bartol et al. 1999; Lenhardt 2002; Ridgway et al. 1969). Sea turtles lack specialized auditory adaptations for higher frequency hearing found in marine mammals. The hearing range for turtles is similar to fish discussed below, in that they have an inner ear capable of detecting low frequency sound. The hearing range for sea turtles was reviewed in a study completed by the (U.S. Navy 2017) to determine the acoustic threshold for this species discussed below.

3.1.8.1.2 Elasmobranchs

Elasmobranchs (giant manta ray, scalloped hammerhead shark, and smalltooth sawfish), like all fish, have an inner ear capable of detecting sound and a lateral line capable of detecting water motion caused by sound (Hastings and Popper 2005; Popper and Schilt 2008). Data for elasmobranch suggest they are capable of detecting sounds from approximately 20 Hz to 2 kHz with the highest sensitivity to sounds at lower ranges (Casper 2006; Casper et al. 2012; Casper and Mann 2007; Ladich and Fay 2013; Myrberg Jr. 2001). However, unlike most teleost fish, elasmobranchs do not have swim bladders (or any other air-filled cavity), and thus are unable to detect sound pressure (Casper et al. 2012). (Popper et al. 2014) also concluded that the risk of mortality, mortal injury, or recoverable injury for fish with no swim bladders exposed to low frequency active sonar was low, regardless of the distance from the sound source.

3.1.8.1.3 ESA-listed Fish

Below is a description of the available information on hearing for these species that was reviewed. There is no available information on the hearing capabilities of Atlantic or shortnose sturgeon specifically, although the hearing of 2 other sturgeon species has been studied. While sturgeon have closed swim bladders, they are unlikely to be used for hearing, and thus sturgeon appear to only rely directly on their ears for hearing. Sturgeon are physostomous, meaning they have a swim bladder that is connected to the gut via a pneumatic duct that allows them to gulp air from the water surface and expel air quickly to adjust the volume of air within the swim bladder. Sturgeon use their swim bladder to actively control buoyancy but do not appear to use it for hearing.

Nassau grouper are physoclistous, meaning they have a gas gland that provides gas exchange by diffusion between the swim bladder and blood. Nassau grouper use gas exchange to regulate their buoyancy and are more vulnerable to barotrauma because they can't pressurize/depressurize as quickly as a sturgeon. Their more developed swim bladder provides greater hearing bandwidth and sensitivity (Ladich and Popper 2004). Halvorsen et al. (2012a); Halvorsen et al. (2012b)suggests that the presence and type of swim bladder correspond with barotrauma injuries at the higher sound exposure levels, with physoclistous fish being more sensitive to higher sound exposure levels than physostomous fish. However, both sturgeon and Nassau grouper are fish and, for the purposes of this Opinion, are considered to have the same hearing range until more information is available to further distinguish how their hearing ranges differ or differences in how they perceive sound. Accordingly, this information does not change our conclusion on the hearing of sturgeon other than that they may be less sensitive to sound that other fish.

Hastings and Popper (2005) reported that studies measuring responses of the ear of European sturgeon (*Acispenser sturio*) using physiological methods suggest sturgeon are likely capable of detecting sounds from below 100 Hz to about 1 kHz. Lake sturgeon (*Acispenser fulvescens*) can detect pure tones from 100 Hz to 2 kHz, with best hearing sensitivity from 100 Hz to 400 Hz (Meyer and Popper 2002). Lovell et al. (Lovell et al. 2005) also studied sound reception and the hearing abilities of paddlefish (*Polyodon spathula*) and lake sturgeon, and determined that lake sturgeon were responsive to sounds ranging in frequency from 100 Hz to 500 Hz, with the lowest hearing thresholds from frequencies in a bandwidth of between 200 Hz and 300 Hz and higher thresholds at 100 Hz and 500 Hz; lake sturgeon were not sensitive to sound pressure. We assume hearing sensitivities for these other species of sturgeon are representative of the hearing sensitivities of all Atlantic sturgeon DPSs and shortnose sturgeon, because sturgeon species are expected to generally respond to sound similarly due to their biological similarities. As previously stated, we assume these same ranges for Nassau grouper, as this species also has a swim bladder, and we therefore expect that it has similar sound thresholds. Popper et al. (2014) concluded that the relative risk of a fish eliciting a behavioral response to a low-frequency sonar was low, regardless of the distance of the sound source. The authors did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking will only occur in a narrow range of frequencies being masked by the sonar transmissions (Popper et al. 2014).

3.1.8.1.4 ESA-listed Whales

NMFS finalized its *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing* in(NMFS 2018a). The Technical Guidance provides the hearing range for whales that may be found in the action area (Table 14). This guidance is used to determine the hearing range of blue, sei, fin, and sperm whales. For North Atlantic right whales, we reviewed another study that considered the anatomical modeling of their hearing range and predicted it to be 10 Hz - 22 kHz with a functional range probably between 15 Hz to 18 kHz (Parks et al. 2007). For this Opinion, we will consider the hearing range of North Atlantic right whale to be 10 Hz to 22 kHz instead of the 7 Hz and 35 kHz predicted for all other low-frequency cetaceans (NOAA 2018).

3.1.8.2 Acoustic Thresholds of ESA-listed Species and how species respond to sound

The point at which sound reaches a level that a species reacts either by causing a behavioral response or by the sound waves injuring the animal is calculated by comparing the level of sound at the distance the animal is from the sound source and to the threshold at which the response is expected for that species. Injury can occur to an animal that is exposed to a sound source that exceeds the peak pressure threshold for that species, which results in an immediate adverse/injurious effect to the animal. Injury can also occur if the animal is experiences prolonged exposure to a sound source that exceeds the daily cumulative sound exposure level (cSEL) threshold for the animal, which results in a physical injury to the animal over a certain time period. Physical injury from exposure to noise can range from minor physiological effects to physiological effects that could potentially result in mortality. Non-fatal injuries are often classified as permanent threshold shifts (PTS) and temporary threshold shifts (TTS). PTS refers to permanent hearing loss or permanent change in hearing thresholds and frequencies and is the irreversible loss of hearing abilities (either total or partial). Permanent threshold shift and other physical injuries could have negative consequences for an animal. TTS refers to a temporary hearing loss or impairments from exposure to loud noises that are recoverable with time (hours to days), neither of which are considered a permanent physical injury, but could have some temporary effects to the auditory sense of an animal and are considered as physical injury when discussing the effects of sound exposure. Behavioral effects are the changes in behavior that occur with the behavioral threshold is exceeded and may include startling responses and changes in foraging or reproductive behaviors.

Determining the threshold at which injury or behavioral effects may occur to a species is dependent on if the sound is impulsive or non-impulsive.

- Impulsive sound: Sound sources that produce sounds that are typically transient, brief (less than 1 second), broadband, and consist of high peak sound pressure with rapid rise time and rapid decay (ANSI 1986; ANSI 1995; NIOSH 1998). They can occur in repetition or as a single event. For this Opinion, only boomers associated with Geophysical surveys are considered an impulsive sound source.
- Non-impulsive sound: Sound sources that produce sounds that can be broadband, narrowband or tonal, brief or prolonged, continuous or intermittent and typically do not have the high peak sound pressure with rapid rise time that impulsive sounds do. For this Opinion, we consider marine vessels operation and aerial surveys to be non-impulsive sound sources. The remaining geophysical sound sources (i.e., chirp sub-bottom profilers, side-scan sonar, multibeam, and single beam echosounders) are also considered non-impulsive sound sources. For non-impulsive sound, cSEL thresholds tend to be lower than peak pressure thresholds.

3.1.8.2.1 Sea turtles Behavioral and Injury Impulsive Noise Thresholds

In order to evaluate the behavioral threshold for exposure to impulsive noise for sea turtles, we reviewed the best available data on the behavioral threshold for sea turtles, which currently comes from studies by O'Hara and Wilcox (1990) and McCauley et al. (2000), who experimentally examined behavioral responses of sea turtles in response to seismic airguns. O'Hara, (O'Hara and Wilcox 1990) found that loggerhead turtles exhibited avoidance behavior at estimated sound levels of 175 to 176 dB re: 1 micropascal (μPa) (rms) (or slightly less) in a shallow canal. (McCauley et al. 2000) reported a noticeable increase in swimming behavior for both green and loggerhead turtles at received levels of 166 dB re: 1 μPa (rms). At 175 dB re: 1 μPa (rms), both green and loggerhead turtles displayed increased swimming speed and increasingly erratic behavior (McCauley et al. 2000). Based on these data, we assume that sea turtles would exhibit a behavioral response when exposed to received levels of 175 dB re: 1 μPa (rms) and higher.

In order to evaluate the thresholds for exposure to impulsive noise by sea turtles that could result in physical effects, we relied on the available literature related to the noise levels that would be expected to result in sound-induced hearing loss (i.e., TTS and PTS); we relied on acoustic thresholds for PTS and TTS (provided in Table 15) for impulsive sounds developed by the U.S. Navy for Phase III of their programmatic approach to evaluating the environmental effects of their military readiness activities (U.S. Navy 2017). At the time of this consultation, we consider this the best available data since the Navy relied on all available information on sea turtle hearing and employed the same methodology to derive thresholds as in NMFS recently issued technical guidance for auditory injury of marine mammals (NMFS 2018a).

Table 15. Acoustic Thresholds for Sea Turtles from Impulsive Sound Sources (U.S. Navy 2017a)

Hearing Group	Generalized Hearing Range	Permanent Threshold Shift Onset (dB re: 1 μPa)	Temporary Threshold Shift Onset (dB re: 1 μPa)
Sea Turtles	30 Hz to 2 kHz	204 cSEL 232 Sound Pressure Level (SPL)(0-peak)	189 cSEL 226 SPL (0-peak)

3.1.8.2.2 ESA-listed Teleost Fish (Sturgeon, Nassau Grouper) and Elasmobranch Behavioral and Injury Noise Thresholds and Sea Turtle Behavioral and Injury Thresholds for Non-Impulsive Noise

NMFS formed a Fisheries Hydroacoustic Working Group in 2004 to evaluate the effects of in-water noise on species under NMFS's jurisdiction from underwater pile driving. Since pile driving is an impulsive sound source, we believe it is appropriate to use the same thresholds for impulsive noise in this Opinion. The Fisheries Hydroacoustic Working Group consists of biologists from NMFS West Coast Region, USFWS, Federal Highway Administration, and the California, Washington and Oregon Department of Transportations, supported by national experts on sound propagation activities that affect fish and wildlife species of concern. In June 2008, the agencies signed a memorandum of understanding documenting interim criteria for assessing physiological effects of pile-driving on fish, until more information was available to reassess these threshold criteria. The criteria were developed for the acoustic levels at which physiological effects to fish could be expected (the onset level), referred to as interim threshold standards below. It should be noted that these onset levels for physiological effects are not necessarily the levels at which fish experience mortality and that some physiological effects are minor and recoverable. Fish are considered more sensitive to physical injury than sea turtles; therefore, fish thresholds are used for both fish and sea turtles as conservative interim criteria and continue to be used for both fish and sea turtles for non-impulsive thresholds.

These interim threshold standards for impulsive noise for fish (inclusive of elasmobranchs) and sea turtle species are used by all of the NMFS regional offices:

- Peak pressure from a single strike: 206 decibels relative to 1 micro-Pascal (dB re 1 μPa).
- cSEL: 187 decibels relative to 1 micro-Pascal-squared second (dB re 1μPa^2-s) for fishes above 2 grams (0.07 ounces).
- cSEL: 183 dB re 1μPa^2-s for fishes below 2 grams (0.07 ounces).

For purposes of assessing behavioral effects of pile driving at several West Coast projects, NMFS has employed a 150 dB re 1 μPa RMS sound pressure level threshold above which it estimates fish species will experience behavioral effects at several sites including the San Francisco-Oakland Bay Bridge and the Columbia River Crossings. This behavioral threshold is also currently used by the NMFS's Greater Atlantic Regional Office and the Southeast Regional Office. The thresholds used by NMFS's Southeast Regional Office are summarized in Table 16 below using the impulsive threshold for fish from the Fisheries Hydroacoustic Working Group. Additional non-impulsive thresholds for fish and sea turtles used by NMFS's Southeast Regional Office from (Halvorsen et al. 2012a; Halvorsen et al. 2012b) are also included in Table 16.

Table 16. Impulsive Acoustic Threshold for ESA-Listed Fish (i.e., sturgeon, Nassau grouper, and elasmobranchs) and Non-Impulsive Acoustic Threshold for Sea Turtles and ESA-listed Fish

Effect	**Threshold Level (dB re 1 μPa)**
Impulsive Sound Source for Fish	
Physical Injury (peak pressure)	206 (peak pressure)
Physical Injury (Cumulative exposure)- Fish under 2 grams	183 cSEL
Physical Injury (Cumulative exposure)- Fish over 2 grams	187 cSEL
Behavior	150 (RMS)
Non-Impulsive Sound Source for Fish and Sea Turtles	
Physical Injury for Sea Turtles and Fish (all sizes)(peak pressure)	206 (peak pressure)
Physical Injury (Cumulative exposure)- Fish under 102 grams	191 cSEL
Physical Injury (Cumulative exposure)- Sea Turtles and Fish over 102 grams	234 cSEL
Behavior for Fish	150 (RMS)
Behavior for Sea Turtles (Section 3.1.8.2.1 of this Opinion)	175 (RMS)

Research continues on the effects of in-water construction on fish, marine mammals, and other aquatic species. Studies by researchers indicate that the threshold of noise impacts to fish may be higher than the interim NMFS thresholds established by the Fisheries Hydroacoustic Working Group in 2008. NMFS is working to review continued research to determine if new guidance and thresholds are warranted that may include weighted thresholds such as those considered for marine mammals. For instance, Halvorsen et al. (2012a) exposed 3 species of fish to pile driving noise in a laboratory setting. Following testing, fish were euthanized and examined for external and internal signs of barotrauma. Halvorsen et al. (2012a) classified the types of fish tested by differences in their anatomy that result in different physiological changes to these groups of fish from pile driving, as described below:

- Fish without swim bladders: Halvorsen et al. (2012a) tested hogchoker fish, a flat bodied fish, for this category of fish. In the action area, species without a swim bladder include elasmobranchs.
- Physostomous (fish with open swim bladders): Halvorsen et al. (2012a) describes these fish as more evolutionarily ancestral than fish with a closed swim bladders (described below). Physostomous fish have a swim bladder that is connected to the gut via a pneumatic duct that allows them to gulp air from the water surface and expel air quickly to adjust the volume of air within the swim bladder. They tested the lake sturgeon in their experiment for this category of fish. ESA-listed physostomous fish in the action area include Atlantic and shortnose sturgeon.

- Physoclistous (fish with closed swim bladders): Halvorsen et al. (2012a) describes these fish as recently evolved fish species. Physoclistous fish have a gas gland that provides gas exchange by diffusion between the swim bladder and blood. They tested Nile tilapia in their study for this category of fish. The only ESA-listed physostomous fish in the action area is the Nassau grouper.

All 3 categories of fish were exposed to a series of 5 trials beginning with a cSEL of 216 to a cSEL of 204 dB re 1 micropascal^2 per second (derived from 960 pile strikes and 186 dB re 1 micropascal^2 per second [sSEL]). In this and subsequent studies by Halvorsen et al. (2012a); Halvorsen et al. (2012b), injuries were categorized by a response weighted index to categorize injuries as mild, moderate, or mortal. The authors defined mild injuries response weighted index 1 as those that were non-life threatening. Based on this and similar studies, the authors made recommendations for cumulative noise exposure thresholds to be raised from the current 187 dB cSEL to 210 dB cSEL, because only mild injuries were observed up to 210 dB cSEL. Because we consider even mild injuries to be physical injury and we are concerned about the potential starting point/onset of physical injury and not the mean of when only mild injuries were observed, NMFS will continue to use the injury thresholds summarized in Table 16 above to be conservative and protective of ESA-listed species, while acknowledging that the cumulative sound exposure threshold may be adjusted as new research becomes available.

One potentially important result of the Halvorsen et al. (2012a) study is that the hogchoker (fish without a swim bladder) did not suffer visible external or internal injuries at lower cSEL levels tested, while the swim bladder fish still suffered mild internal injuries. The fish with open swim bladders also suffered fewer internal injuries than the fish with closed swim bladders. As more research continues, this may lead to policy changes for different species of ESA-listed fish such as elasmobranchs that also lack a swim bladder. Although, another consideration for bottom-dwelling elasmobranchs such as the smalltooth sawfish is that they are often in contact with the substrate (Casper et al. 2012). The vibrations within the sediment from sound waves could also be damaging, especially when considering the body shape of sawfish. Many of the organs of these dorsoventrally flattened fishes are in close proximity to the bottom surface of the body, providing little protection from pile-driving vibrations. It is unclear if the Halvorsen et al. (2012a) study took into consideration the secondary effects of noise from vibrations within the sediment.

3.1.8.2.3 ESA-listed Whale Behavioral and Injury Noise Thresholds

Marine mammals are continually exposed to many sources of sound. Naturally occurring sounds such as lightning, rain, sub-sea earthquakes, and biological sounds (e.g., snapping shrimp, whale songs) are widespread throughout the world's oceans. Marine mammals produce sounds in various contexts and use sound for various biological functions including, but not limited to social interactions, foraging, orientation, and predator detection. Interference with producing or receiving these sounds may result in adverse impacts. Audible distance, or received levels of sound depend on the nature of the sound source, ambient noise conditions, and the sensitivity of the receptor to the sound (Richardson et al. 1995). Type and significance of marine mammal reactions to sound are likely dependent on a variety of factors including, but not limited to, the behavioral state of the animal (e.g., feeding, traveling, etc.), frequency of the sound, distance between the animal and the source, and the level of the sound relative to ambient conditions (Southall et al. 2007).

For this analysis of effects to whales (blue, fin, sei, sperm, and North Atlantic right whale), we will refer to information provided in the NMFS *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing* (NMFS 2018a). The Technical Guidance is a document that compiles, interprets, and synthesizes scientific literature to produce updated acoustic thresholds to assess how anthropogenic, or human-caused, sound affects the hearing of all marine mammals under NOAA Fisheries jurisdiction. These acoustic thresholds cover the onset of both TTS and PTS. Acoustic thresholds refer to the levels of sound that, if exceeded, will likely result in temporary or permanent changes in marine mammal hearing sensitivity. Additionally, the current NMFS approach is to consider the behavioral threshold for marine mammals to be 160 dB re 1micropascal (RMS) for all geophysical survey equipment types considered under this Opinion. The peak pressure and cumulative thresholds considered for whales in this Opinion are provided in Table 17. This analysis does not consider the effects to marine mammals under the MMPA.

Table 17. Acoustic Threshold for Marine Mammals (NMFS 2018a)

Hearing Group	**Impulsive Sound Source (dB re: 1 micropascal)**	**Non-Impulsive Sound Source**
Low-frequency cetaceans (Blue, fin, sei, and North Atlantic right whale)	219 peak 183 dB cSEL (24 hour)	199 cSEL (24 hour)
Mid-frequency cetaceans (Sperm whale)	230 peak 185 cSEL (24 hour)	198 cSEL (24 hour)

3.1.8.3 Sound Generated from the Operation of Equipment

The information available on the sound generated by the operation of equipment covered under this Opinion varies by study and is generally focused on vessel traffic and dredging. Vessel traffic studies considered various types of vessels including those covered under this Opinion and the acoustic effects of dredging were considered separately since the sound generated by machinery penetrating the substrate results in additional acoustic effects beyond just those of the vessel. As discussed above, sound generated by activities related to dredging and material placement is non-impulsive. Sound from geophysical surveys, which is considered impulsive, is addressed separately below in Section 3.1.8.4 of this Opinion.

A recent paper *Evaluating Effects of Dredging-Induced Underwater Sound on Aquatic Species: A Literature Review* (USACE 2019a) compared available acoustic studies. The literature review found that the sounds generated by dredging are low frequencies under 1,000 Hz (Reine et al. 2014; Reine and Dickerson 2014). Sound decibel levels recorded from various dredging types range from approximately 100 -190 dB re 1 μPa at 1 m (Dickerson et al. 2001; Thomsen et al. 2009). Geotechnical survey equipment was measured to have a source levels of 142–145 dB re 1 μPa rms @ 1 m and a frequency range of 30–2000 Hz for drilling using a 20 kW, 83 mm diameter drillbit, 1500 rpm, 16–17 m drill depth in sand and mudstone (Erbe and McPherson 2017). The vibratory mechanism on the vibracorer would introduce sound into the substrate and some sound would be radiated into the water column as well. The vibratory mechanism produces a relatively short, broadband noise with peak frequency less than 1 kHz. Source levels are generally expected to be less than 180-190 dB re 1 μPa @ 1 m depending on the intensity of the vibrations, barrel material, and nature of sediment penetration (Reiser et al. 2010). We believe that the sound generated during geotechnical surveys covered under this Opinion will be similar or less due to the expected much smaller size drill needed to take sediment samples covered under this Opinion, which is typically 4 inches. Based on this information, we believe that sound generated by equipment operated under this Opinion will not result in a peak pressure injury since all sound source levels measured at 1 m from the source are under the threshold of 206 dB peak for sea turtles and fish (Table 16). No non-impulsive peak pressure threshold is provided for marine mammal injury from non-impulsive sounds, because non-impulsive sounds are not characterized by high peak sound pressures and rapid rise times (NMFS 2018a). Therefore, no injury is expected.

Another study reviewed the cumulative sound exposure of commercial shipping and dredging by hopper dredging in the Netherlands and found that the cumulative levels were 182 dB cSEL (24 hour exposure) for shipping vessels and 186 dB cSEL (24-hour exposure) for shipping and hopper dredging combined (USACE 2019a). Based on this information, we believe the continuous exposure to noise from dredging and vessels operating on projects covered under this Opinion will result in no injurious acoustic effects to ESA-listed species as a result of cumulative sound exposure. The threshold for cumulative sound effects from non-impulsive sound sources is 234 dB cSEL for sea turtles and fish over 102 grams and 191dB cSEL for fish under 102 grams (Table 16), both of which are higher than the recorded 186 cSEL dB during the Netherlands acoustic study, which we consider to be the best available scientific information. Similarly, the threshold for marine mammals from non-impulsive cumulative sound exposure is 199 dB cSEL (24 hour exposure) for low-frequency cetaceans and 198 dB cSEL (24 hour exposure) for high-frequency cetaceans (Table 17), both of which are higher than the 186 cSEL dB during the Netherlands acoustic study.

Dredging and vessels operated for projects covered under this Opinion can result in behavioral effects to ESA-listed species since they will operate at a frequency within the hearing range of ESA-listed species and exceed the behavioral threshold. Based on the available information, it is not clear how much the behavioral threshold is exceeded. Studies show that much of the increase in sound in the ocean environment is due to increased shipping, as vessels become more numerous and of larger tonnage (Hildebrand 2009; McKenna et al. 2012; National Research Council 2003). Commercial shipping continues to be a major source of low-frequency sound in the ocean, particularly in the Northern Hemisphere where the majority of vessel traffic occurs. Although large vessels emit predominantly low frequency sound, studies report broadband sound from large cargo vessels above 2 kHz. The low frequency sounds from large vessels overlap with many ESA-listed species predicted hearing ranges (7 Hz-35 kHz) (NMFS 2018a) and may mask their vocalizations and cause stress (Rolland et al. 2012). The broadband sounds from large vessels may interfere with important biological functions of marine mammals, including foraging (Blair et al. 2016; Holt 2008). At frequencies below 300 Hz, ambient sound levels are elevated by 15 to 20 dB when exposed to sounds from vessels at a distance (McKenna et al. 2013). Analysis of sound from vessels revealed that their propulsion systems are a dominant source of radiated underwater sound at frequencies less than 200 Hz (Ross 1976). Additional sources of vessel sound include rotational and reciprocating machinery that produces tones and pulses at a constant rate. Measurements made over the period 1950 through 1970 indicated low-frequency (50 Hz) vessel traffic sound in the eastern North Pacific Ocean and western North Atlantic Ocean was increasing by 0.55 dB per year (Ross 1976; Ross 1993; Ross 2005). While we acknowledge that the noise generated by vessel traffic is of concern, we expect that any behavioral effects from noise exposure from activities contemplated in this Opinion will be insignificant and that mobile species would be able to avoid areas of behavioral sound disturbances while work is occurring and return once it is complete. We also considered if the sound generated would affect important biological functions including feeding, sheltering, and reproduction and determined that the PDCs in this Opinion limit activities that would result in adverse effects to these functions. We believe that the behavioral effects will be insignificant and will not alter any important biological functions since these species are mobile and can move away from these sound sources and continue to use similar habitat in surrounding areas. Mobile species are expected to be able to avoid sound generated, if disturbed, and return once completed; however, as discussed in Section 3.1.1 of this Opinion, sturgeon do not seem to avoid vessel and construction noise and do not exhibit behavioral responses in their presence that indicate they alter an important life function in response to sound generated by this type of equipment. Of greatest concern is masking of communication of ESA-listed species. While Nassau grouper are known to communicate in aggregation areas, these areas are outside of the action area and will not be affected. Sea turtles, elasmobranchs, and other ESA-listed fish in the action area are not known to audibly communicate and thus their ability to communicate would not be affected by this continuous sound source. ESA-listed whales in the area are predominately located off of the continental shelf in waters deeper than where work covered under this Opinion would occur and would not be expected to be present in areas affected by noise from vessels and dredging covered under this Opinion, except North Atlantic right whales. However, North Atlantic right whales appear to not communicate with calves when in calving areas and therefore any masking of communication from vessel traffic is expected to be insignificant. Additionally, this Opinion will not result in an increase in vessel traffic.

3.1.8.4 Sound generated from geophysical survey

We used information available from multiple sources to determine sound sources and calculate distance those sounds would travel for geophysical equipment covered under this Opinion. Much of this came from information provided by BOEM for NMFS consultations on similar geophysical equipment including the information provided in Table 18 -Table 20. This information was then used to determine the effects to ESA-listed species. BOEM engages frequently in these activities, and scientific literature on noise effects from geophysical surveys is limited. We accordingly consider this information to be the best scientific and commercial information available.

Table 18. Geophysical Survey Sound Characteristics[32]
(Final Environmental Assessment, Sand Survey Activities for BOEM's Marine Minerals Program, Atlantic and Gulf of Mexico, Appendix A, Table A-1 (BOEM 2019))

Source	Operational Frequency Range	Peak Source Level	Representative Beam Pattern (Horizontal and Vertical)	Frequency for Testing	Representative Pulse Length (ms)	RMS SPL (dB)	SEL (dB)	Cumulative SEL (dB) 10 seconds
Boomer (surface tow)	300 Hz - < 10 kHz	< 220 dB	Horizontal: omnidirectional Vertical: downward focused	6.2 kHz	0.06	205	172	182
Chirp subbottom profiler (tow above seafloor)	500 Hz -24 kHz	< 220 dB	Horizontal: omnidirectional Vertical: downward focused	4-20 kHz (wideband)	4.6	180	156	166
Side-scan sonar (near-surface tow)	> 180- 900 kHz	< 240 dB	Along-track: very narrow Across-track:wide	445 kHz	0.1	223	182	192
Multibeam (hull or davit mounted)	> 180- 500 kHz	< 230 dB	Determined by number of beams, beam spacing, frequency, etc. Along-track: very narrow Across-track:wide	300 kHz	0.3	221	185	195
Interferometric Swath (davit mounted)	> 180- 600 kHz	< 220 dB	Depends on frequency Along-track: very narrow Across-track: wide	234 kHz	0.2	218	180	190
Single Beam (hull mounted)	> 180- 540 kHz	< 230 dB	Horizontal: omnidirectional Vertical: Downward	200 kHz	0.7	194	163	173

[32] Shading indicates the operational frequency is beyond hearing range of cetaceans, manatees, seals, sea turtles, and most fish.P1PTesting by Crocker and Fratantonio 2016; see report for details. All source levels are in dB re 1 µPa @ 1 m, while SEL values are in dB re 1 µPaP2P s.

Table 19. Calculated Distance to Level A and B Harassment for Marine Mammals
(Final Environmental Assessment, Sand Survey Activities for BOEM's Marine Minerals Program, Atlantic and Gulf of Mexico Appendix A, Table A-1 (BOEM 2019)[33])

	Range to Level B (m)		**Range to Level A (m)**			
Source	**All Marine Mammals 160 dB**	**Low-frequency Cetaceans (199 dB)**	**Mid-frequency Cetaceans (198 dB)**	**High-frequency Cetaceans (173 dB)**	**Phocid Pinnipeds (201 dB)**	**Otariid Pinnipeds (219 dB)**
Boomer (surface tow)	192 @ 205 dB 89 @ 200 dB	< 1	< 1	< 3	< 1	< 1
Chirp sub-bottom profiler (tow above seafloor)	7 @ 180 dB	< 1	< 1	< 1	< 1	< 1

Table 20. Geophysical Survey Sound Characteristics (BOEM 2014) tested by JASCO[34]

Source	**Operational Frequency Range**	**Peak Source Level**	**Representative Beam Pattern (Horizontal and Vertical)**	**Representative Pulse Length (ms)**	**JASCO - Modeled Maximum Distance to rms SPL 160dB[35]**	**JASCO Modeled Maximum Distance to rms SPL 160dB[36]**	**JASCO Observed Distance to rms SPL 160 dB[37]**	**JASCO Observed Distance to rms SPL 160 dB[38]**
Boomer (surface tow)	300 Hz - < 10 kHz	< 220 dB	Horizontal: omnidirectional Vertical: downward focused	< 1	1 - 2.1 km [212 dB re 1 µPa at 1 m, 0.2-16 kHz]	< 50 m [206 dB re 1 µPa at 1 m, 0.2- 16 kHz]	12 m [SL unknown, 300 Hz – 14 kHz]	--
Chirp sub-bottom profiler	500 Hz ─ 24 kHz	< 220 dB	Horizontal: omnidirectional	10-50	0.35 – 1 km [222 dB re 1 µPa at 1	< 40 m [210 dB re 1 µPa at 1 m, 2-16 kHz]	10 m [210 dB re 1 µPa at 1 m, 2-16 kHz]	30 m – 80 m [210 dB re 1

[33] Only sources with operational frequencies (peak and primary) within hearing range of cetaceans, manatees, seals, sea turtles, and most fish reported.
[34] Distance to Received Level from Representative Sources Source Level (SL), Operational Frequency for all JASCO modeling.
[35] Source: (BOEM 2014), Appendix D. Zykov and Carr 2012
[36] Source: Zychov, M. 2013. Underwater Sound Modeling of Low Energy Geophysical Equipment Operations. JASCO Document 00600, Version 2.0. Technical report by JASCO Applied Sciences for CSA Ocean Sciences Inc.
[37] Source: Martin, B., J. MacDonnell, N. E. Chorney, and D. G. Zeddies. 2012. Sound Source 20 Verification of Fugro Geotechnical Sources: Final Report: Boomer, Sub-Bottom 21 Profiler, Multibeam Sonar, and the R/V Taku. JASCO Document 00413, Version 1.0 DRAFT. Technical report by JASCO Applied Sciences for Fugro GeoServices Inc. 31 pp.
[38] Source: Zychov, M., and J. MacDonnell. 2013. Sound Source Characterizations for the Collaborative Baseline Survey Offshore Massachusetts Final Report: Side Scan Sonar, Sub-Bottom Profiler, and the R/V Small Research Vessel experimental. JASCO Document 00413, Version 2.0. Technical report by JASCO Applied Sciences for the (US) Bureau of Ocean Energy Management.

Source	Operational Frequency Range	Peak Source Level	Representative Beam Pattern (Horizontal and Vertical)	Representative Pulse Length (ms)	JASCO - Modeled Maximum Distance to rms SPL 160dB[35]	JASCO Modeled Maximum Distance to rms SPL 160dB[36]	JASCO Observed Distance to rms SPL 160 dB[37]	JASCO Observed Distance to rms SPL 160 dB[38]
(tow above seafloor)			Vertical: downward focused		m, 3.5, 12, 200 kHz]			µPa at 1 m, 0.5-12 kH
Side-scan sonar (near-surface tow	> 180– 900 kHz Frequency above hearing range of cetaceans, manatees, seals, sea turtles, and most fish.	< 240 dB	Along-track: very narrow Across-track: wide	< 0.5	500 – 650 m [226 dB re 1 µPa at 1 m, 100/400 kHz]	< 700 m [234 dB re 1 µPa at 1 m, 132/500kHz]	--	--
Multibeam (hull or davit mounted)	> 180 - 500 kHz Frequency above hearing range of cetaceans, manatees, seals, sea turtles, and most fish.	< 230 dB	Determined by number of beams, beam spacing, frequency, etc. Along-track: very narrow Across-track: wide	< 0.5	150 m [173.5 dB re 1 µPa2·s at 1 m (SEL), 240 kHz]	< 300 m [221 dB re 1 µPa at 1 m, 200/400kHz]	1 m [SL unknown, 200/400 kHz]	--
Interferometric Swath (davit mounted)	> 180 – 600 kHz Frequency above hearing range of cetaceans, manatees, seals, sea turtles, and most fish.	< 220 dB	frequency Along-track: very narrow Across-track: wide	< 0.5	--	--	--	< 10 - 20 m [SL unknown, 234 kHz]
Single Beam (hull mounted)	> 180 – 540 kHz Frequency above hearing range of cetaceans, manatees, seals, sea turtles, and most fish	< 230 dB	Horizontal: omnidirectional Vertical: downward	0.1	--	<30 m [230 dB re 1 µPa at 1 m, 200 kHz]	2 m [Source level unknown, 70 /200 kHz	--

3.1.8.4.1 Effects Boomer and Chirp Sub-Bottom Profilers

Boomer and chirp sub-bottom profilers are the only 2 geophysical survey equipment types that will be operated within the hearing range of ESA-listed species since the G&G PDCs in Appendix G limit all other geophysical surveying to occur above 160 kHz.

We believe that there will be no injurious effect to ESA listed species from boomers and chirp sub-bottom profilers. The G&G PDCs in Appendix G state that geophysical surveys will be limited to operating under 205 dB source level for boomers and chirp sub-bottom profilers (G&G PDCs in Appendix G). As stated in Table 18, if these equipment types are operated at less than 220 dB source level, the peak pressure is 172 dB (SPL [source peak level]), which is less than the 206 dB (SPL) injury threshold for sea turtles and fish (including elasmobranchs; see Section 3.1.8.2 of this Opinion) and less than the threshold for PTS onset of 219 dB for low frequency cetaceans and 230 dB for mid-frequency cetaceans (Table 17).

We believe that there also will be no injurious effect to ESA listed species from cumulative noise exposure from boomers and chirp sub-bottom profilers, which are an impulsive noise source. As stated in Table 18, if these equipment types are operated at less than 220 dB source, the 10 second cumulative exposure is 182 dB (cSEL) for boomers and 166 dB (cSEL) for chirp sub-bottom profilers, which is less than the 183 dB (cSEL) cumulative exposure threshold for sea turtles and fish over 2 grams, 183 dB (cSEL) and for fish under 2 grams (Table 16), 189 dB calculated for sea turtle TTS (Table 15), 183 dB PTS for 24 hour exposure to low-frequency cetaceans and 185 dB PTS for 24 hour exposure to mid-frequency cetaceans (Table 17). Table 19 also calculated the Level A harassment limits for boomers and chirp sub-bottom profilers if the sound source was 205 dB for a boomer and determined that the radius of the Level A harassment is less than 1 m for low and mid-frequency cetaceans. Under the MMPA, Level A harassment means any act of pursuit, torment, or annoyance that has the potential to injure a marine mammal or marine mammal stock in the wild. This is similar to how we categories physical injury for ESA-listed species when considering acoustic effects. In order to be exposed to potentially injurious levels of noise from a sound source 1 m away, the risk of injury would be from direct physical contact with the vessel or geophysical survey equipment. We believe that mobile species will be able to avoid directly encountering these survey equipment types and avoid injury.

We believe that the behavioral effects to ESA-listed whales from boomer and chirp sub-bottom profilers will be insignificant. Boomer and chirp sub-bottom profiles will be operated at a source level under 205 dB to meet the G&G PDCs in Appendix G. At 205 dB, the radius that the sound would remain above 160 dB (the threshold for marine mammal behavioral effects) is about 192 m (Table 18). Impulsive noise is more disruptive for an animal's behavioral response that non-impulsive noise because it is startling and unpredictable. However, since geophysical survey equipment is towed through an area, the area where increased underwater noise occurs will be small and transient with the chance of a particular area being ensonified lasting only a few seconds. We expect any behavioral effects from sound generated to be minor and limited to a temporary disruption of normal behaviors or temporary avoidance of the ensonified area. Of greatest concern would be if behavioral effects masked communication between whales or resulted in a change to a species during an important life stage. To that end, geophysical surveying that is within the hearing range of North Atlantic right whales is not allowed when they may be present to prevent behavioral effects to them while in the action area calving. All other ESA-listed whales are expected to be rare in the areas where geophysical surveys will occur, since they typically occupy deeper water areas off the continental shelf where these activities will not occur. Therefore, we think exposure to these whales from a directional sound source pointed below the geophysical surveying equipment is extremely unlikely.

We believe that behavioral effects to sea turtles, sturgeon, Nassau grouper, and elasmobranchs from boomer and chirp sub-bottom profilers will be insignificant. These species may be present in areas where surveying occurs and able to detect the sound source within 192 m; however, if exposed to this mobile sound source, the exposure would only last a few seconds. Any temporary disturbance is not expected to meaningfully change behavioral and these species would be able to resume activities after the survey vessel passes. Unlike whales, sea turtles, sturgeon, Nassau grouper, and elasmobranchs in the area are unlikely to have these sound sources mask their communication. In particular, Nassau grouper aggregation areas are outside of the action area and will not be affected. Sea turtles, elasmobranchs, and other ESA-listed fish in the action area are not known to audibly communicate and thus their ability to communicate would not be affected by this continuous sound source. Also, these species are not expected to be deterred from important life functions from a short exposure to this sound source.

3.1.8.4.2 Effects from Side-Scan Sonar, Single Beam Echosounder, Multibeam Echosounder, and Interferometric

We believe that there will be no injurious effect to ESA listed species from side-scan sonar, single beam echosounder, multibeam echosounders, and interferometric survey equipment. These are intermittent non-impulsive sound sources that involve geophysical survey equipment dragged through an area, with small ensonified zones. For non-impulsive sound sources, which are not characterized by high peak pressure levels, the thresholds for cumulative sound exposure (cSEL) tend to be lower than the peak thresholds, as indicated in Table 16. The effect from these non-seismic sound sources outside of the hearing range of ESA-listed fish and sea turtles is limited. Calculations of the radius of impact tend to be overly conservative relative to field observations (Table 20), since they do not take into account sound transmission loss of high frequency sound sources through the water, which are known to dissipate more quickly. In general, observations suggest a radius of impact to 160 dB of 500 m for side-scan sonar, 2 m for single-beam echosounder, 1 m for multibeam echosounder, and 10-20 m for interferometric (Table 20); however 160 dB will likely result in larger isopleths compared to the cSEL thresholds in Table 16. For example, in Table 16, 191 cSEL is presented as the cumulative exposure threshold for fish under 10 g (the most vulnerable group listed), while the 160 dB threshold is not cumulative. Since there is a 31 dB difference between 160 dB and 191 dB cSEL, this means that an animal would need to be exposed to a sound source for ~1259 seconds at 160 dB to accumulate enough sound to exceed the 191 dB cSEL threshold (cSEL = RMS + 10 log duration in seconds). Furthermore, all these sound sources have pulse lengths less than 1 second. Table 18 provides the cSEL for 10-second exposures from these sources, with only two of these sources exceeding the 191 dB cSEL threshold (i.e,. multibeam and sidescan sonar). For a multibeam sonar, with a pulse length of 0.3 seconds, this means that it would take 33 pulses to result in 10 seconds of exposure. Hence the likelihood of an individual being exposed to all 33 pulses is low. If these sources are moving and do not produce omnidirectional sound (i.e., have a beamwidth), the likelihood of exposure is reduced even more. Thus, for any sound source, the cSEL isopleth is likely smaller than the 160 dB isopleth. Additionally, these intermittent non-impulsive sound sources have not been observed to result in instantaneous injury, and these species are mobile and therefore unlikely to be subject to a prolongued exposure. Finally, PSOs will be monitoring for ESA-listed species. Thus, we believe that exposure to peak pressure from these equipment types for sufficient time to cause injury is extremely unlikely. Similarly, we believe that the risk from cumulative sound exposure is low because these are moving sound sources with directional surveying, meaning that it is extremely unlikely that an animal would be exposed to this noise for a prolonged period of time. For elasmobranchs, Popper et al. (2014) concluded that the risk of mortality, mortal injury, or recoverable injury for fish with no swim bladders like elasmobranchs exposed to low frequency active sonar was low, regardless of the distance from the sound source supporting the risk of injury from these equipment types to be discountable.

Since smaller fish are more susceptible to injury from cumulative sound exposure, we also specifically considered the risk of injury to juvenile Nassau grouper and sturgeon in spawning rivers. We expect that there will be no effect to Atlantic or shortnose sturgeon since work covered under this Opinion is limited to portions of the rivers not used for spawning and that by the time that sturgeon traveled downriver from spawning areas to portions of the river where work may occur that these sturgeon would be larger than the 102 gram size limit for additional cumulative sound exposure concerns. Similarly, Nassau grouper of this smaller size class are limited to nearshore areas in the U.S. Caribbean, so we believe that the risk of exposure is so low that we consider this route of effect to be discountable. Surveying conducted under this Opinion is not expected to occur in these areas and the radius of exposure from these sound sources is small and would extend into nearshore areas where juvenile grouper are expected to be when of this size class.

We believe that behavioral effects to ESA- listed species from side-scan sonar, single beam echosounder, multibeam echosounder, and interferometric survey equipment are extremely unlikely, and therefore believe that this route of effect is discountable. The G&G PDCs in Appendix G do not include projects that operate these equipment types at frequencies within the hearing range of ESA-listed species (specifically they must not operate under 160 kHz). If a sound source falls outside the hearing range of a species (or harmonics are only detectable at much lower levels), it is considered inaudible and so a behavioral disturbance is considered extremely unlikely. We specifically considered impacts to marine mammals, as they are particularly sensitive to sounds (NMFS 2018a); however, marine mammals are not known to be injured by sound sources outside of their hearing range so injurious effects to ESA-listed whales are also extremely unlikely.

3.1.8.5 Noise generated from aerial survey noise

The effects to marine mammals from aerial surveys was recently analyzed in a NMFS Biological Opinion addressing research on marine mammals in the Atlantic Ocean (FPR-2018-09256 and FPR-2018-09257). In this Opinion, NMFS concluded that the presence and noise generated during aerial surveys using manned aircraft would have an insignificant effect on North Atlantic right whales. We reach the same conclusion here. We expect that the surveys will result in no reaction or only mild short-term behavioral reactions and will not result in any long-term behavioral changes or reduction in fitness. Cetacean (whales, dolphins, and porpoise) responses to aircraft depend on the animals' behavioral state at the time of exposure (e.g., resting, socializing, foraging, or traveling) as well as the altitude and lateral distance of the aircraft to the animals (Luksenburg and Parsons 2014). The underwater and sound intensity from aircraft is less than produced by boats and visually, aircraft are more difficult for cetaceans to locate since they are not in the water and move rapidly (Richter et al. 2006). However, when aircraft fly below certain altitudes (about 500 m [1,640.4 ft] as will occur for aerial surveys considered under this Opinion), they have caused cetaceans to exhibit behavioral responses that might constitute a significant disruption of their normal behavioral patterns (Patenaude et al. 2002). Thus, aircraft flying at low altitude, at close lateral distances and above shallow water elicit stronger responses than aircraft flying higher, at greater lateral distances and over deep water (Patenaude et al. 2002; Smultea et al. 2008a; Smultea et al. 2008b). The sensitivity to disturbance by aircraft may also differ among species (Wursig et al. 1998). Sperm whales have been observed to respond to a fixed-wing aircraft circling at altitudes of 245 to 335 m (803.8-1,099.1 ft) by ceasing forward movement and moving closer together in a parallel flank-to-flank formation, a behavioral response interpreted as an agitation, distress, and/or defense reaction to the circling aircraft (Smultea et al. 2008a; Smultea et al. 2008b). About 14% of bowhead whales approached during aerial surveys exhibited short-term behavioral reactions (Patenaude et al. 2002). While all ESA-listed whale species exposed to aerial surveys may exhibit short-term behavioral reactions, data from the Northwest Fisheries Science Center and Northeast Fisheries Science Center from past permits indicated only mild behavioral responses, if any. This was compared to records of observed responses to North Atlantic right whale aerial surveys conducted under the 1997 SARBO and determined to be the same. We assume that the response of other ESA-listed whales would be similar or less than those observed of right whales and it is less likely that the other ESA-listed whale species (sperm, blue, sei, fin) will be exposed to aerial survey sounds since the surveys they are typically found further off shore that where these surveys are flown. Therefore, we believe that the exposure to sound from aerial surveys covered under this Opinion will have an insignificant effect on ESA-listed species.

3.2 Potential Routes of Effects to Non-Mobile ESA-Listed Species

This Section analyzes the routes of effects of the proposed action to non-mobile ESA-listed species that may occur within the action area (Johnson's seagrass and ESA-listed corals listed in Table 8). In particular, Section 3.2.1 of this Opinion analyzes Johnson's seagrass interaction with vessels and equipment (Section 3.2.1.1), potential direct removal or burial (Section 3.2.1.2), and potential effects from water quality changes (Section 3.2.1.3). Section 3.2.2 of this Opinion analyzes potential routes of effects to ESA-listed corals in the action area, including: species interactions with vessels and equipment (Section 3.2.2.1), potential direct removal (Section 3.2.2.2), and potential effects from turbidity, sedimentation, or burial (Section 3.2.2.3). Routes of effect identified as leading to adverse effects on these species are discussed further in Section 6.

3.2.1 Johnson's Seagrass

Johnson's seagrass is limited in range to shallow water protected areas in southeast Florida (as described in the Johnson's seagrass PDCs in Appendix D) such as Biscayne Bay and the IWW. Areas deeper than 13 ft are assumed to be too deep to support adequate water transparency for Johnson's seagrass growth. Studies show that Johnson's seagrass occurs in waters shallower than 10-13 ft (3-4 m) (NMFS 2007a). Water depths greater than 13 ft are not believed to provide the water transparency necessary for sufficient sunlight to reach the sea floor to support Johnson's seagrass growth and are therefore not considered to be within the range of Johnson's seagrass for the analysis in 2020 SARBO.

3.2.1.1 Species Interaction with Vessels and Equipment

Equipment placement within the range of Johnson's seagrass will avoid areas with any seagrasses, to the maximum extent practicable, including turbidity curtain anchors, barge spudding or anchoring, pipelines, or other materials. Vessels will also follow deep water channels to avoid potential groundings that could damage Johnson's seagrass (Section 2.2 in Appendix B). In cases where pipeline placement cannot avoid seagrass, floating pipelines will be used instead of anchored pipelines to prevent placement on Johnson's seagrass. While equipment staged in a dredged channel may be placed on Johnson's seagrass, the loss of seagrass in those areas will be evaluated under the dredging activities that will be discussed further in Section 6.2 of this Opinion. We believe that the placement of all other equipment allowed under this Opinion will have insignificant effects to Johnson's seagrass since areas with Johnson's seagrass will be avoided. Any temporary placement of equipment on Johnson's seagrass would occur rarely and be temporary in nature. Therefore, we would expect such placement to have an insignificant effect, as plants would be expected to continue to persist following the removal of equipment. Additionally, equipment placement in areas without this species will not permanently alter the habitat. Geotechnical surveys are not covered under this Opinion if seagrasses are present, pipelines will be floated over seagrass beds, and equipment will be preferentially staged in the channel where work is occurring.

3.2.1.2 Direct Removal or Burial from Dredging or Material Placement

The Johnson's seagrass PDCs limit dredging, material placement, and the equipment that will be used for these activities, to minimize the risk of removal of Johnson's seagrass from dredging or equipment placement (see above), habitat alteration or burial from material placement or sedimentation. While dredging is limited to maintaining existing dredge footprints (e.g., channels and other already dredged locations), Johnson's seagrass frequently recruits into recently dredged areas including maintained navigation channels. In fact, observations by researchers have suggested that Johnson's seagrass exploits unstable environments or newly-created unvegetated patches by exhibiting fast growth and support for all local ramets in order to exploit areas in which it could not otherwise compete. It may quickly recruit to locally uninhabited patches through prolific lateral branching and fast horizontal growth. While these attributes may allow it to compete effectively in periodically disturbed areas, if the distribution of this species becomes limited to stable areas it may eventually be outcompeted by more stable-selected plants represented by the larger-bodied seagrasses (Durako et al. 2003). Therefore, dredging those areas, even maintenance dredging of routinely dredged areas, will result in adverse effects to Johnson's seagrass if it is in the channel. The loss of Johnson's seagrass from dredging will be disused further in Section 6.2 of this Opinion.

We believe that there will be no effect to Johnson's seagrass from borrow area dredging within the range of Johnsons' seagrass as the PDCs limit this activity within the range of Johnson's seagrass to only one routinely used site (Baker's Haulover). The Baker's Haulover borrow site is at an inlet in an area with high currents and no aquatic vegetation. Therefore, work occurring in this area will not result in direct burial or sedimentation covering seagrass in the area. All other borrow area dredging will occur at sites deeper than 13 ft deep (i.e., beyond the depth limit where Johnson's seagrass is expected). Dredging or material placement in waters too deep to support the growth of Johnson's seagrass are not expected to affect Johnson's seagrass as the PDCs require that the work occurs a sufficient distance from seagrasses that turbidity generated from the work does not result in sedimentation on the adjacent seagrasses.

Material placement within the range of Johnson's seagrass is not covered under this Opinion (e.g., nearshore placement, side-cast, ODMDS [which does not occur within the range of Johnson's seagrass]) and therefore will have no effect on Johnson's seagrass from burial of the seagrass by the material or turbidity generated from the activity.

3.2.1.3 Water Quality Changes from Dredging and Material Placement

Johnson's seagrass may also be affected if it is growing adjacent to the channel, from turbidity generated during dredging resulting in sedimentation landing on or burying adjacent seagrasses; however, we believe those effects will be insignificant. Having a canopy of only 2 centimeter (cm) -5 cm, this species may be easily covered by sedimentation and mesocosm experiments indicate that clonal fragments can only survive burial for up to a period of 12 days (W.J. Kenworthy, NOAA's Center for Coastal Fisheries and Habitat Research, Beaufort, North Carolina, 1997 unpublished). For this reason, the Johnson's seagrass PDCs limit dredging within the range of Johnson's seagrass to suction type dredging such as cutterhead and hopper dredging, which generally have lower levels of turbidity, and limit mechanical dredging with the use of turbidity curtains, limit overflow from hopper dredging or other equipment such as scows to only areas that are at least 500 ft from Johnson's seagrass, and limit water injection dredging to only areas that are 1,000 ft from Johnson's seagrass (Johnson's seagrass PDCs in Section 2 in Appendix D). We believe that these measures will ensure that effects from turbidity generated will be sufficiently minimized to prevent burial of this species. However, the distance that turbidity and sedimentation may travel in areas where Johnson's seagrass may be found from these equipment types is based on limited information. Therefore, the Johnson's seagrass PDCs require pre-construction surveys to determine in Johnson's seagrass is growing near dredging areas and post-construction surveys to determine if the PDC minimization measures effectively protected the seagrasses. The effectiveness of these minimization measures will be discussed during the SARBO annual review to ensure that the insignificant effects determination remains accurate (Section 2.9 of this Opinion).

3.2.2 ESA-listed Corals

The 7 ESA-listed coral species present in the action area occur on shallow coral reefs throughout the Wider-Caribbean, including south Florida, Puerto Rico, and the U.S. Virgin Islands within the action area. Individual species have different, but mostly overlapping ranges and occupy different reef environments and depths (Table 21). In this Opinion, we rely on the presence of habitat in which corals grow (referred to as coral hardbottom[39]) to determine if ESA-listed corals may be present, as ESA-listed corals will only occur in these areas. We do not have, and it would be impracticable to obtain, coral surveys of the entire action area. Additionally, because ESA-listed corals are sessile and rely on coral hardbottom to recruit and survive, we analyze impacts to both the coral colonies and to their ability to use and access their habitat, coral hardbottom, for each route of effect below.

Table 21. ESA-Listed Coral Species Distribution within the SARBO Action Area.

Coral Species	Reef Environment	Depth Distribution	U.S. Geographic Distribution within SARBO Action Area
Staghorn *(Acropora cervicornis)*	spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hardbottom habitats	most common 5-25 m total range 0-30 m	Southeast Florida from the northern limit of Palm Beach County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands
Elkhorn *(Acropora palmata)*	fore-reef, reef crest, and shallow spur-and-groove zone	most common 0.5-5 m total range 0.5-40 m	Southeast Florida from the northern limit of Broward County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands
Lobed star *(Orbicella annularis)*	most reef environments	most common 1-10 m total range 1-80 m	Southeast Florida from Lake Worth Inlet in Palm Beach County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands
Mountainous star *(Orbicella faveolata)*	most reef environments	most common 10-20 m total range 1-80 m	Southeast Florida from St. Lucie Inlet in Martin County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands
Boulder star *(Orbicella franksi)*	most reef environments	most common 15-30 m total range 5-80 m	Southeast Florida from Lake Worth Inlet in Palm Beach County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands
Pillar *(Dendrogyra cylindrus)*	most reef environments	most common 5-15 m total range 1-25 m	Southeast Florida from Lake Worth Inlet in Palm Beach County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands
Rough cactus *(Mycetophyllia ferox)*	most reef environments	most common 10-20 m total range 5-30 m	Southeast Florida from Broward County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands

3.2.2.1 Species Interaction with Vessels and Equipment

3.2.2.1.1 Vessels and equipment placement (other than pipelines)

We believe that, with the exception of pipeline placement (addressed separately below), the placement of equipment and use of vessels for projects covered under this Opinion will have no effect to ESA-listed corals or coral hardbottom due to the conditions in the Coral PDCs in

[39] For purposes of the 2019 SARBO, coral hardbottom is defined in the same way as the essential feature for *Acropora* critical habitat: as substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. "Substrate of suitable quality and availability" is defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover.

Appendix C. The Coral PDCs identify the conditions under which equipment will be placed in relation to coral hardbottom, which are protective of ESA-listed coral colonies and its habitat (coral hardbottom). Equipment placement within the range of ESA-listed corals will avoid areas with any coral hardbottom, and therefore areas near ESA-listed coral colonies, including vessel anchoring and spudding (PDC Coral.1). Vessels will also follow deep water channels to avoid potential groundings that could damage coral (required in the General PDCs in Section 2.2 in Appendix B).

3.2.2.1.2 Pipelines

Pipelines used to transport dredged material to beaches for beach nourishment projects within the range of ESA-listed corals will only be placed in routinely used pipeline corridors selected to avoid coral hardbottom and corals. This Opinion does not cover the use of new pipeline corridor locations. Pipelines are placed on the seafloor in a manner to ensure that they do not move and to ensure they do not cause damage to adjacent habitat by ensuring the pipeline is of sufficient size, weight, or anchored to prevent movement. In cases where pipeline placement crosses areas with coral hardbottom, the pipeline will be floated over the hardbottom or placed on risers to prevent placement of the pipeline on ESA-listed corals.

While we expect that most projects will not result in any impacts to ESA-listed corals or coral hardbottom, the USACE has proposed as part of the proposed action to relocate ESA-listed corals if damage could occur to ESA-listed corals from the placement of a pipeline. The Coral PDCs in Appendix B describe how surveys along the pipeline would be performed to determine if coral hardbottom is in the pipeline path, if ESA-listed corals had regrown within the pipeline corridor, and the project-specific consultation/coordination with NMFS that will occur prior to any coral relocation, and how to relocate corals when deemed necessary. As further fiscussed in Section 6.3 of this Opinion, we expect that these instances will be rare, as only existing pipeline corridors are covered by this Opinion, and these pre-existing corridors were selected to avoid impacts to corals and coral hardbottom. Additionally, any ESA-listed corals there would be limited to those that had regrown since the last use of the pipeline corridor. Based on the limited locations of pipeline corridors within the range of ESA-listed corals and a review of information regarding these corridors (NOAA. National Centers for Coastal Ocean Science (NCCOS), unpublished data), we believe that staghorn, elkhorn, slobed star, and mountainous star corals could be present and need to be relocated. The potential of adverse effects to these corals from relocation are discussed in Section 6.3 of this Opinion. We assume no pillar or rough star coral colonies occur in pipeline corridors because these corals, already found in low densities in the action area, are declining due to disease events in the action area, and it is extremely unlikely that one of these species would colonize within a previously used pipeline corridor. Additionally, USACE has not previously found these corals during pipeline projects similar to those that will occur under 2020 SARBO. Therefore, we believe encountering pillar or rough star coral colonies during pipeline placement is extremely unlikely, and therefore consider this route of effect to be discountable.

This Opinion also considers the potential effects if the pipeline cannot be floated and must be placed on coral hardbottom or placed on risers on hardbottom to suspend the pipeline (typically 6-12 inches above the hardbottom), if the pipeline moves after it is placed resulting in damage to ESA-listed corals or coral hardbottom, or if the pipeline has a structural failure resulting in an unexpected blowout of sand. The Coral PDCs in Appendix B describe how surveys along the pipeline would be performed to monitor for these events. The USACE has no reports of a pipeline fail resulting in a blowout during past projects in the action area and we assume based on the rarity of this event that it will not happen under this Opinion and that there will be no effect to ESA-listed corals or coral hardbottom from a pipeline blowout. The pipeline pressure will be monitored closely so that any loss of pressure would be detected prior to a blowout that could damage ESA-listed corals or coral hardbottom. However, it is possible that a pipeline could move, especially during a storm. In the unlikely event of pipeline movement, all dredging, pumping, and filling will stop until corrective actions are taken to secure the pipeline to prevent further movement. Post-construction surveys will be conducted to determine if damage occurred; however, we believe that the temporary placement of risers on hardbottom or if pipeline movement did occur, would not cause permanent damage to coral hardbottom, given its consolidated nature. Therefore, we believe that this habitat would still be able to support corals. Additionally, we do not expect that ESA-listed corals would be present within the pipeline corridor when the pipeline is placed, as any corals within the pipeline corridor would be relocated, as discussed above. Therefore, the effects to corals resulting from physical interactions with pipelines, or due to habitat alteration by pipeline placement would be insignificant. The relocation of coral colonies from pipeline corridors is discussed in Section 6.3.3.

3.2.2.1.3 Dredging

We believe there will be no effect to ESA-listed corals, or coral hardbottom, from direct removal by dredging. Dredging covered under this Opinion is limited to sand mining in borrow sites where corals are not expected to occur or maintenance of existing navigation channels. Maintenance dredging of navigation channels does not include deepening or widening the channel, which would penetrate hard substrate along either the bottom or sides of the channel. Since corals require hardbottom to grow, we do not expect them to occur in the areas along the bottom of the channel where accumulated sediments will be removed. Even if sections of the bottom of the channel do not have an accumulation of sediment at the time of dredging, we expect that they would not support coral growth due to the depth of the channel and sediments in the channel. The PDCs do not require surveys for corals in the channel (on the bottom or the sidewalls). While corals are not expected in navigation channels for the reasons described above, they can grow along the sidewalls of navigation channel. However, this Opinion does not cover widening of channels and therefore these corals will not be removed.

3.2.2.1.4 Beach Nourishment

We believe that beach nourishment covered under this Opinion will have no effect on ESA-listed corals, or coral hardbottom, by direct removal or burial within the authorized fill template. The Coral PDCs in Appendix C limit beach nourishment within the range of ESA-listed corals to the previously filled ETOF within the previously authorized beach fill template. We do not expect ESA-listed corals to occur in any hardbottom areas that may be exposed since a previous beach nourishment event. Given the low sexual recruitment rates of ESA-listed corals, time between nourishment events is likely too short to allow ESA-listed corals to settle and grow. Hardbottom exposed within the previously filled beach template is not expected to be functioning as habitat that supports ESA-listed corals due to the limited time it is exposed and that it is likely within a high energy wave zone that would not support corals. This Opinion does not cover projects that place sand on coral hardbottom in areas not previously constructed.

We also expect that beach nourishment projects filled to the previously filled beach template ETOF that are filled with beach quality sand (as required by the PDCs) will generally not result in sedimentation or burial of ESA-listed species or hardbottom outside of the ETOF. The original beach fill template is designed to limit sand placed on the beach from extending beyond the limits of the ETOF. However, in limited instances, sedimentation resulting from beach nourishment may affect ESA-listed corals or coral hardbottom outside of the previously filled ETOF. The potential for adverse effects from sedimentation related to beach nourishment are discussed below, in Section 3.2.2.2 and Section 6 of this Opinion.

3.2.2.1.5 Relocation Trawling

We believe that there will be no effect to ESA-listed corals from relocation trawling. Trawling is used to relocate mobile ESA-listed species by dragging a net through an area to capture them and move them out of the way of harm from hopper dredging. However, relocation trawling can also result in damage to bottom habitat. Therefore, the General PDCs in Section 3.5 of Appendix B prohibit relocation trawling in areas within the range of ESA-listed corals, unless considered under Section 2.9.5.1 of this Opinion, to ensure that trawling will only be done in areas where coral do not occur. Additionally, the PDCs do not cover relocation trawling in the U.S. Caribbean.

3.2.2.2 Turbidity, Sedimentation, and Burial of ESA-listed Corals and Coral Hardbottom

ESA-listed corals may be impacted by turbidity caused by dredging and material placement. The major problem arising from turbidity is related to shading caused by decreases in ambient light. Suspended sediments, especially when fine-grained, decrease the quality and quantity of light levels, resulting in a decline in photosynthetic productivity of the zooxanthellae that live within coral tissue and provide them with nutrition (Richmond 1993). High turbidity rates may depress coral growth and survival due to attenuation of light available to symbiotic zooxanthellae. High turbidity can also reduce successful fertilization, larval survival, and larval settlement, resulting in decreased reproduction (Gilmour 1999).

While sediment movement and deposition is a normal process in a coral reef ecosystem, the amount of sedimentation that a coral and coral hardbottom can be exposed to before causing harm is limited. A study of offshore coral reefs concluded that corals are not capable of developing or sustaining biological or ecological functions when covered by more than a centimeter of sediment for prolonged periods of time (Erftemeijer et al. 2012). Sedimentation can also reduce coral settlement and survival of recruits, and NMFS has previously determined that sediment depths of 0.5 cm (or more) of fine sediment precludes coral recruitment and fragment attachment (NMFS 2016b). The potential effects of sediment input not only include direct mortality, but also sublethal effects such as lower calcification rates and reduced productivity, reduced or ceased reproduction, bleaching, physical damage to coral tissue and reef structures (e.g., abrasion), and reduced regeneration from tissue damage. Sedimentation on reefs can reduce coral recruitment, survival, and settlement of larvae, suppress colony growth (Bak 1978), and may increase disease prevalence (Pollock et al. 2014). Fine (silts and clays) or non-native sediments can be more damaging to corals than coarse or native reef sediments, especially to recruits and juveniles (Fourney and Figueiredo 2017). Adult corals will attempt to clean themselves of sediment using a combination of ciliary action and the production and sloughing off of mucus sheets. These actions use a great deal of energy and can lead to exhaustion of mucus-producing cells (Riegl 1995; Riegl and Branch 1995). Energy diverted to clearing the colony surface of sediment can lead to growth inhibition and a reduction in other metabolic processes (Dodge and Vaisnys 1977; Rogers 1983). At the population level, increased sedimentation may inhibit sexual recruitment, change the relative abundance of species, decrease live coral cover, and reduce the abundance and diversity of corals (Gilmour 1999; Rogers 1990).

3.2.2.2.1 Dredging

We believe turbidity and sedimentation generated during dredging covered under this Opinion will have an insignificant effect on ESA-listed corals. The Coral PDCs (Section 2 of Appendix C) limit dredging equipment, duration, sand composition, and proximity of work to coral hardbottom to minimize the risk and impact of effects to ESA-listed corals and their habitat coral hardbottom. The PDCs provide specific limitations for the dredging to be covered under this Opinion within the range of ESA-listed corals based on the presence of fines sediment (fines) and proximity to ESA-listed corals and coral hardbottom. The PDCs limit dredging within 1,000 ft of coral hardbottom to sediments with less than 10% of fines, limit the equipment that can be used to dredge these areas to those expected to produce less turbidity, and limit the dredging duration to 18 days. Additionally, overflow by hopper dredging or other support equipment will not be allowed within 1,000 ft of coral hardbottom if fines are 5-10%.

The limitation on the amount of fine particles in the dredged material is based on our understanding of the distance the turbidity can travel and result in sedimentation and burial effects to corals in the area. Sand has a particle size that makes it heavy compared to fine silt and clay materials, causing sand to resettle on the bottom quickly after being re-suspended. In a laboratory study exposing coral recruits to sediment with 20% fines, survival dropped off dramatically after 3 weeks of exposure, though survival dropped off sooner at higher concentrations of fines coupled with higher temperatures (Fourney and Figueiredo 2017). Nelson et al. (Nelson et al. 2016) developed a stoplight indicator system to predict the conditions under which corals exposed to dredging would be expected to recover. Based on a number of studies examining the effects of sedimentation on corals, they determined that minimal stress to corals when the total sedimentation was less than 0.5 cm, which occurred with a deposition rate of less than 10 mg/centimeter^2(cm^2)/day and suspended sediment concentration less than 10 mg/L in the bottom 2 m for 18 days in any 90-day running window. NMFS has previously determined that sediment depths of 0.5 cm (or more) of fine sediment precludes coral recruitment and fragment attachment meaning that the habitat would no longer be functioning as designated critical habitat (NMFS 2016b). We assume sedimentation under 0.5 cm to be an insignificant effect, as sedimentation movement is a natural part of ocean systems and therefore should dissipate following the completion of a project. The Coral PDCs Appendix C limit the duration of dredging within 1,000 ft of areas where corals may be present (coral hardbottom) to less than 18 days, and only allow dredging near corals and coral hardbottom when sediment has 10% fines or less, as stated above. Based on these limitations, we believe exposure of ESA-listed corals and coral hardbottom to sedimentation from dredging will be minimized and result in less than 0.5 cm deposition and minimal stress to ESA-listed corals to a point that the sedimentation will have an insignificant effect on ESA-listed corals. We also expect that any impacts to ESA-listed corals' ability to use coral hardbottom for settlement and recruitement purposes will be insignificant, because the PDCs limit dredging within 1,000 ft of coral hardbottom to sediments with less than 10% of fines, limit the equipment that can be used to dredge these areas to those expected to produce less turbidity, and limit the dredging duration to 18 days. Additionally, overflow by hopper dredging or other support equipment will not be allowed within 1,000 ft of coral hardbottom if fines are 5-10%.

The PDCs provide fewer restrictions for dredging within the inner port/harbor areas of Port Everglades, Miami Harbor, and San Juan Harbor, which are within the range of ESA-listed corals. ESA-listed corals are not expected to be present in these inner port/harbor areas that will be maintenance dredged, as the bottom of the channel would not support coral recruitment. However, ESA-listed corals and coral hardbottom have been identified in areas immediately outside of the inner port/harbor locations to be dredged (locations of these inner port/harbor locations are defined in the Coral PDCs). Dredging in these inner port/harbor areas have different requirements under the PDCs and allow dredging of sediments with higher fines, do not have time restrictions, and allow other equipment types to be used. We evaluated the risk of turbidity generated during this dredging in the ports/harbors away from corals resulting in sedimentation and burial of corals outside of the port or harbor. Based on the USACE reports of its previous dredging experience in these areas (Pers Comm, USACE staff to NMFS, Alison Moulding), we do not expect dredging sediments with higher than 10% of fines in these semi-enclosed port/harbor areas to travel to corals or coral hardbottom outside of the port/harbor, and therefore believe that dredging in these areas will have an insignificant effect on ESA-listed corals and corals' ability to use nearby hardbottom habitat. Fine-grain sediments, arising from land based sources, are retained within the semi-enclosed portions of the harbors/ports due to the hydrology in these areas resulting in the settling on sediments in specific spots within the ports/harbor that later need to be maintenance dredged. We accordingly do not expect that fine sediment dredged in these areas will result in the same impacts expected if such fines were dredged in an open ocean environment previously discussed. However, if new information becomes available that does not support the conclusion that fine-grained sediment dredged within these ports/harbors do not affect nearby ESA-listed corals and coral hardbottom areas, re-initiation of the consultation will be required.

We believe that turbidity generated by dredging in areas near ESA-listed corals that are growing on vertical surfaces will have an insignificant effect on the corals. ESA-listed corals may be found on the vertical walls in navigation channels, berths, pilings, and seawalls, where sediment is not likely to bury them due to their position on a vertical surface that would not accumulate sediment.

3.2.2.2.2 Dredge Material Placement

We believe that turbidity generated during beach nourishment projects within the range of ESA-listed corals may have an adverse effect on ESA-listed corals), if sediment extends beyond the ETOF, and ESA-listed corals are within range of sedimentation. The Coral PDCs in Appendix C limit placement of sand in the same beach template as was previously filled. While we generally do not expect ESA-listed corals or coral hardbottom outside of the previously filled ETOF to be buried if filled to the same beach fill template, there may be adverse effects in limited instances, which are discussed here and further in Section 6 of this Opinion. Beaches will be nourished using beach-quality sand, which, as stated in the PDCs has a low percentage of fines (≤ 10%). The low percentage of fines in combination with the use of construction methods to minimize turbidity, such as shore parallel dikes to allow settling of sand, is expected to result in low suspension of fine materials that can cause turbidity and affect water quality. As discussed in Section 3.2.2.1.4 of this Opinion, ESA-listed corals and coral hardbottom are not expected to occur within the previously filled beach template. While the original beach fill template determined the ETOF as the waterward limit that sand would extend from the shoreline, turbidity and sedimentation could extend beyond that limit in limited circumstances.

The Coral PDCs in Appendix C require that surveys be completed prior to a beach nourishment project within the range of ESA-listed corals to determine if coral hardbottom or ESA-listed corals occur within 500 ft of the ETOF. If present, the Coral PDCs allow ESA-listed corals within 500 ft to be relocated, if NMFS is contacted pursuant to Section 2.9.5.1 of this Opinion to confirm that turbidity and sedimentation generated during the beach placement would result in adverse sedimentation on these adjacent corals. The requirement to survey out to 500 ft from the ETOF is to verify what corals or coral hardbottom might be present, if specific site conditions exist that could lead to turbidity and sedimentation within the area, and if any ESA-listed corals should be relocated to avoid sedimentation effects. We believe that if turbidity and sedimentation is a concern to adjacent corals, it would be limited to within 100 ft of the ETOF, based on past experience. Any coral relocation would be coordinated with NMFS prior to relocation to determine if it is necessary. The effects of relocating these corals is analyzed further in Section 6.3 of this Opinion. As discussed in Section 6.3, we do not believe that Pillar or Rough Cactus corals will be present in areas where beach nourishment will occur.

Temporary sedimentation on coral hardbottom in areas adjacent to the ETOF is expected to be minimal and temporary and would naturally be reduced or removed by wave action and would therefore have an insignificant effect on corals through effects to coral hardbottom. We believe that the beach-quality sediment with low fine composition will settle out of the water column within the ETOF and result in less than 0.5 cm sediment deposition on coral hardbottom outside of the ETOF, due to the dynamic nature of the nearshore environment. Thus, even in areas where ESA-listed corals may be relocated to minimize the risk of lethal take, we expect that sedimentation on coral hardbottom would resolve quickly as a result of natural processes.

3.2.2.2.3 Pipelines

We believe there will be no effect to ESA-listed corals or coral hardbottom from turbidity and sedimentation from a pipeline, as discussed above in Section 3.2.2.1 of this Opinion.

3.3 Potential Routes of Effects to Critical Habitat from Projects Covered under this Opinion

As discussed at the beginning of Section 3 of this Opinion, this section will discuss the expected effects to designated critical habitat from activities covered under this Opinion. In Section 2 of this Opinion, the activities analyzed under this Opinion were divided into 5 general categories (listed below).

1. Dredging (including all forms of dredging discussed in Section 2.3). For the effects analysis below, geotechnical surveys (discussed in Section 2.6.4) will be considered within the scope of the effects analysis for mechanical dredging since this activity removes material by taking samples of sediment, though geotechnical survey material removal is at a much smaller scale.
2. Placement of dredged materials (including all forms discussed in Section 2.4). Placement in an upland location is outside of NMFS jurisdiction and will have no effect on designated critical habitat under NMFS jurisdiction. Therefore, upland placement will not be included in the analysis of each critical habitat below.
3. Equipment used for dredging, transportation, and material placement (discussed in Section 2.5).
4. Geophysical surveys authorized by the USACE necessary to complete dredging and material placement projects (discussed in Section 2.6.3). Since geophysical surveys are limited to dragging equipment through the water column that emits a sound used to survey the sea floor, we believe that this category of activity will have no effect to any of designated critical habitat. This activity does not contact the sea floor or alter the water column in any way, except to generate sound, which will not affect the physical or biological features of any critical habitat. Therefore, this activity will not be included in the analysis of each critical habitat below.
5. Monitoring for and handling of ESA-listed species encountered during projects covered under this Opinion (discussed in Section 2.7).

Past critical habitat designations have used the terms primary constituent element (PCE) or essential feature (EF) to identify important habitat qualities. However, the new critical habitat regulations (81 FR 7413; Publication Date February 11, 2016) replace those terms with physical or biological features (PBF). This shift in terminology does not change the approach used in conducting our analysis, whether the original designation identified PCE, EF, or PBF. For simplicity, we universally apply the term PBF in this Opinion for all critical habitat, regardless of the term used in the specific critical habitat designation. Critical habitat boundary maps are available at https://www.fisheries.noaa.gov/tags/southeast-critical-habitat-map.

Our effects analysis for each critical habitat discussed below begins with a general description of where the critical habitat is designated within the action area and the PBF(s) of each designated critical habitat. Some of the designated critical habitat units within the action area include areas that were excluded from the relevant designation of critical habitat, which are also listed at the beginning of each critical habitat analysis, if this information is relevant to the analysis. The areas in which these activities can occur are described in the Action Area Section (Section 2.8 of this Opinion) and are limited by the PDCs (Appendix B - Appendix H).

3.3.1 *Acropora* Critical Habitat

The action area of this Opinion encompasses all areas designated as *Acropora* critical habitat to protect elkhorn and staghorn coral, which is located in both southeast Florida and the U.S. Caribbean. The PBF of *Acropora* critical habitat is provided in Table 22 below.

Table 22. *Acropora* Critical Habitat PBFs

Acropora (Staghorn and elkhorn coral)(73 FR 72210, Publication Date: November 26, 2008)	The physical feature essential to the conservation of elkhorn and staghorn corals is: substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. "Substrate of suitable quality and availability" is defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover.

As discussed in Section 1.1.1 of the Coral PDCs in Appendix C, the following areas are not considered critical habitat and activities occurring in these areas will have no effect on critical habitat. It is important to note that areas not considered critical habitat may still have effects to the species if they occur in this area, as was discussed in Section 3.2.2 of this Opinion.

1. All areas subject to the 2008 Naval Air Station Key West Integrated Natural Resources Management Plan.
2. All areas containing existing (meaning constructed at the time of this critical habitat designation) federally authorized or permitted man-made structures such as aids-to-navigation (ATONs), artificial reefs, boat ramps, docks, pilings, maintained channels, or marinas.
3. All waters identified as existing (already constructed) federally authorized channels and harbors as follows: (i) Palm Beach Harbor; (ii) Hillsboro Inlet; (iii) Port Everglades; (iv) (v) Miami Harbor; (vi) Key West Harbor; (vii) Arecibo Harbor; (viii) San Juan Harbor; (ix) Fajardo Harbor; (x) Ponce Harbor; (xi) Mayaguez Harbor; (xii) St. Thomas Harbor; and (xiii) Christiansted Harbor.

3.3.1.1 Vessels and Equipment

As discussed in Section 3.2.2.1 of this Opinion, we believe that equipment placed during construction will have no effect on the essential feature of *Acropora* critical habitat, except for the placement and use of pipelines used to transport sand to beach nourishment projects. Pipelines used to transport dredged material to beaches for beach nourishment projects are placed in routinely used pipeline corridors selected to avoid essential feature of *Acropora* critical habitat. Pipelines are placed on the seafloor in a manner to ensure that they do not move to ensure they do not cause damage to adjacent habitat by ensuring the pipeline is of sufficient size, weight, or anchored to prevent movement. In cases where pipeline placement crosses areas with essential feature of *Acropora* critical habitat, the pipeline will in most cases be floated over the hardbottom or placed on risers to prevent placement on the essential feature of *Acropora* critical habitat.

While we expect that most projects will not result in any impacts to the essential feature of *Acropora* critical habitat, this Opinion considers the potential effects if the pipeline cannot be floated and must be placed on coral hardbottom, if the pipeline moves after it is placed resulting in damage to ESA-listed corals or coral hardbottom, or if the pipeline has a structural failure resulting in an unexpected blowout of sand. The Coral PDCs in Appendix B describe how surveys along the pipeline would be performed to determine if the essential feature of *Acropora* critical habitat are in the pipeline path, to check for pipeline blowouts, and to check if the pipeline moved. The USACE has no reports of a pipeline fail resulting in a blowout during past projects in the action area and we assume based on the rarity of this event that it will not happen under this Opinion and effects to the essential feature of *Acropora* critical habitat from a pipeline blowout are extremely unlikely to occur. The pipeline pressure will be monitored closely so that any loss of pressure would be detected prior to a blowout that could damage the essential feature of *Acropora* critical habitat. However, it is possible that a pipeline could move, especially during a storm. In the unlikely event of pipeline movement, all dredging, pumping, and filling will stop until corrective actions are taken to secure the pipeline to prevent further movement. Post-construction surveys will be conducted to determine if damage occurred; however, we believe that if pipeline movement did occur, it would not cause permanent damage to the essential feature of *Acropora* critical habitat, due to its consolidated nature, and that this habitat would still be able to support corals following any pipeline movement. Similarly, we expect that placement of a pipeline on coral hardbottom, in the event that the pipeline cannot be floated above hardbottom, will be a temporary effect to *Acropora* critical habitat, because the feature will continue to function immediately following the removal of the pipeline. Therefore, the effects to coral hardbottom would be insignificant.

3.3.1.2 Dredging

Acropora critical habitat may be affected by sedimentation from dredging and material placement (which is limited to beach nourishment projects within *Acropora* critical habitat where the essential features may be present). Sediment accumulation on dead coral skeletons and exposed hard substrate reduces the amount of available substrate suitable for coral larvae settlement and fragment reattachment. Even small increases in sedimentation can significantly reduce coral recruitment and survivorship (Babcock and Smith 2000), and sediments coupled with turf algae further impede recruitment (Birrell et al. 2005). Last, survivorship of branching coral fragments is significantly affected by the type of substrate, with increased mortality being linked to the presence of sandy sediments (Lirman 2000). As stated above, NMFS has previously determined that sediment depths of 0.5 cm (or more) of fine sediment precludes coral recruitment and fragment attachment meaning that the habitat would no longer be functioning as designated critical habitat (NMFS 2016b). We assume sedimentation under 0.5 cm to be an insignificant effect, as sedimentation movement is a natural part of ocean systems and therefore should dissipate following the completion of a project.

We believe dredging activities covered under this Opinion will have no effect to the essential feature of *Acropora* critical habitat from the direct removal of features by dredging. The Coral PDCs (Section 2 of Appendix C) prohibit dredging that requires the penetration of rock or other hard substrate and because maintenance dredging in navigation channels covered under this Opinion does not include any deepening or widening of the existing channel that would require the removal of hard substrate. Therefore, dredging will not occur in areas where the essential feature of *Acopora* critical habitat is present. In addition, channel dredging within the range of corals is limited to mostly navigation channels excluded from critical habitat. Therefore, dredging will have no effect on the essential feature of *Acropora* critical habitat by direct removal of features.

We believe dredging activities covered under this Opinion will have an insignificant effect to the essential feature of *Acropora* critical habitat from turbidity generated during dredging. Dredging may affect nearby *Acropora* critical habitat if turbidity generated resulted in burial or sedimentation of coral hardbottom over 0.5 cm. The Coral PDCs in Appendix C limit dredging to sediments with a low percentage (< 10%) of fines, limit the equipment that can be used to dredge these areas to those expected to produce less turbidity, and limit dredge duration to 18 days in areas within close range of coral hardbottom (e.g., within the distance that we expect sediments to travel via turbidity associated with a project). The amount of fine material in the dredged sediment is used to determine how far turbidity may travel from the dredged location. Sand has a particle size that makes it heavy compared to fine silt and clay materials, causing sand to resettle on the bottom quickly after being re-suspended, while fines take longer to settle and therefore may travel longer distances in the water. The PDCs constraints on dredging, including sediment composition, duration, equipment type, and distance from hardbottom, will minimize the distance sediment is transported and the amount of sediment deposited on coral hardbottom as discussed above in Section 3.2.2.2.1 of this Opinion. We believe deposited sediment will be less than 0.5 cm deep and will be an insignificant effect, to the essential feature of *Acropora* critical habitat.

We believe dredging allowed in the inner port/harbor areas of Port Everglades, Miami Harbor, or San Juan Harbor covered under this Opinion will have an insignificant effect to the essential feature of *Acropora* critical habitat that are located outside of the port/harbor. As discussed in Section 3.2.2.2.1 of this Opinion, we do not expect dredged sediments with higher percentage of fines in the inner areas of the Port Everglades, Miami Harbor, or San Juan Harbor to travel to adjacent *Acropora* critical habitat outside of the port/harbor. These fine-grained sediments settle within the semi-enclosed portions of the harbors/ports due to low water movement. However, if new information becomes available that does not support the conclusion that fine-grained sediment dredged within these ports/harbors do not affect the essential feature of *Acropora* critical habitat, re-initiation of the consultation will be required.

3.3.1.3 Dredge Material Placement

We believe beach nourishment covered under this Opinion will have an insignificant effect to the essential feature of *Acropora* critical habitat. Beach nourishment is limited to previously authorized and constructed beach fill templates as described in the Coral PDCs in Appendix C. This Opinion does not cover projects that place sand on coral hardbottom in areas not previously constructed, and does not include beach nourishment in the U.S. Caribbean. While hardbottom areas may emerge between nourishment events, we do not consider hardbottom areas within the ETOF that have previously been nourished and are potentially re-exposed between nourishment events to be functioning *Acropora* critical habitat. We believe the movement and deposition of sand within these high-energy zones close to shore precludes recruitment of *Acropora* species. Placement of sand in areas not previously filled is not covered under this Opinion. The essential feature of *Acropora* critical habitat may be affected if sediment deposited on hardbottom areas impacts areas outside of the ETOF either as material is placed or as sediment erodes from nourished beaches over time. While sediments may accumulate on corals in limited instances (discussed in Section 3.2.2.2 of this Opinion), we expect the effects of this sediment on the essential feature to be temporary as sand deposition and movement occurs naturally in nearshore areas. Even in areas where ESA-listed corals may be relocated from outside of the existing ETOF to minimize the risk of lethal take, we expect that sedimentation on the essential feature in that same area would resolve quickly as a result of natural processes. Therefore, we believe that material placement for beach nourishment will have insignificant effects on the essential feature of *Acropora* critical habitat. If new information reveals that sedimentation due to beach nourishment results in depths of 0.5 cm or greater of fine sediment over hardbottom within previously unconstructed beach fill templates or outside of the ETOF, re-initiation of the consultation will be required.

3.3.2 Atlantic Sturgeon Critical Habitat

Atlantic sturgeon critical habitat was designated in known spawning rivers along the Atlantic coast of the United States. Within the action area of this Opinion, the designated critical habitat includes rivers in North Carolina, South Carolina, Georgia, and the St. Mary’s River at the Florida/Georgia border. The spawning rivers within the action area are specifically identified in the Sturgeon PDCs in Appendix E and the Atlantic sturgeon PBFs are provided in Table 23 below.

Table 23. Atlantic Sturgeon Critical Habitat PBFs

Atlantic sturgeon (82 FR 39160, Publication Date: August 17, 2017; Effective Date: September 18, 2017)	The physical features essential for the conservation of Atlantic sturgeon belonging to the Carolina and SA DPSs are those habitat components that support successful reproduction and recruitment. These are: (1) Hardbottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0-0.5 parts per thousand range) for settlement of fertilized eggs and refuge, growth, and development of early life stages;
	(2) Aquatic habitat inclusive of waters with a gradual downstream salinity gradient of 0.5 up to as high as 30 parts per thousand and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development; (3) Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support: (i) Unimpeded movement of adults to and from spawning sites; (ii) Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and (iii) Staging, resting, or holding of subadults or spawning condition adults. Water depths in main river channels must also be deep enough (at least 1.2 m) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river; (4) Water quality conditions, especially in the bottom meter of the water column, with temperature and oxygen values that support: (i) Spawning; (ii) Annual and inter-annual adult, subadult, larval, and juvenile survival; and (iii) Larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and oxygen values will vary interdependently, and depending on salinity in a particular habitat. For example, 6.0 mg/L DO or greater likely supports juvenile rearing habitat, whereas DO less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25°C. In temperatures greater than 26°C, DO greater than 4.3 mg/L is needed to protect survival and growth. Temperatures of 13 to 26 °C likely support spawning habitat.

3.3.2.1 PBF 1. Hardbottom Substrate in Low Salinity Water

We believe that there will be no effect to this PBF from any activity covered under this Opinion. The Sturgeon PDCs limit activities to areas of the river downstream of where sturgeon spawning is believed to occur. This protects the habitat needed for spawning as described in PBF 1 and its ecological function (settlement of fertilized eggs, refuge, growth, and development of early life stages). For projects covered under this Opinion, the upstream limits of where work can occur in sturgeon spawning rivers are provided in Table 56 of the Sturgeon PDCs in Appendix E.

Atlantic sturgeon spawning occurs well upstream of where activities under this Opinion will occur, at or near the fall line of rivers, over hard substrate consisting of rock, pebbles, gravel, cobble, limestone, or boulders (Gilbert 1989a; Smith and Clugston 1997). Hard substrate is required so that highly adhesive Atlantic sturgeon eggs have a surface to adhere to during their initial development and young fry can utilize the interstitial spaces between rocks, pebbles, cobble, etc., to hide from predators during downstream movement and maturation (Gilbert 1989a; Smith and Clugston 1997). Additionally, spawning needs to occur in very low salinity (i.e., 0.0 – 0.5 parts per thousand) because exposure to even low levels of salinity can kill Atlantic sturgeon during their first few weeks of life. Thus, larval sturgeon tend to inhabit upstream reaches of rivers for several weeks to months, limiting their downstream movement until they can endure brackish waters (Bain et al. 2000). For PBF 1 to fulfill its role in supporting the conservation objective, not only must hardbottom substrate be present in freshwater, but the location of the substrate must be far enough upstream that larval fish are not exposed to toxic salinity levels too soon.

Table 56 provides the areas (river miles) in sturgeon rivers where work is covered under this Opinion and work is not covered above the upper river miles listed in the table. The upper work limits are based on the best available information to demarcate the locations within each Atlantic sturgeon critical habitat unit above which spawning habitat capable of fulfilling its intended function/ecological role is likely to occur. While it is possible for some activities covered under this Opinion to occur in freshwater, potentially over hardbottom substrate, those features would be located so far downstream from known sturgeon spawning areas that the hardbottom feature would not be used to support spawning functions. Accordingly, these downstream areas would not be considered to be functioning critical habitat supporting PBF 1. Thus, actions there will not affect this PBF of critical habitat. Additionally, and as discussed further below, the proposed action will not affect salinity levels in riverine areas. If new information becomes available, Table 56 will be updated to continue to protect spawning habitat, as described in the Sturgeon PDCs and PDC modification in Section 2.9.5.3 of this Opinion. Activities may occur above this upstream limit on a project by project basis under the Supersede procedures outlined in Section 2.9.5 of this Opinion if sufficient evidence is available that the project will not alter spawning habitat or effect spawning sturgeon, eggs, or other early life stages of sturgeon.

3.3.2.2 PBF 2. Soft Substrate with a Gradual Downstream Salinity Gradient

Dredging projects considered under this Opinion could alter PBF 2 (gradual salinity gradient and soft substrate); however, we believe that there will be no effect to the gradual salinity gradient of the PBF for activities covered under this Opinion. When a channel is deepened and lengthened, the denser saltwater sitting underneath the less-dense freshwater can move further upriver. Increased salinity can change sturgeon prey assemblages and shorten the distance available for animals to undergo physiological changes to prepare for salinity exposure. However, this Opinion only covers dredging in previously authorized dredge templates and no new deepening or widening of channels is covered under the PDCs. Thus, the salinity range and salt water wedge in each of these rivers will not change from what has already occurred during the original deepening or widening.

As discussed in more detail in Section 3.1.7 of this Opinion under the analysis regarding habitat alteration in sturgeon spawning rivers above, activities covered under this Opinion will alter soft sediment used by Atlantic sturgeon for foraging; however, we believe the effects to the soft substrate described in PBF 2, and therefore effects to PBF 2, will be insignificant. As sturgeon hatch, grow, and move downstream toward estuarine habitats, it is essential for developing fish to forage in areas of soft substrate and to encounter gradual changes in salinity to allow physiological adaptations to higher salinity waters. Thus, this feature is necessary for juvenile foraging and physiological development. Since the soft substrate that will be altered by activities covered under this Opinion contains sturgeon prey, alteration of soft substrate is expected to temporarily remove or reduce the availability of that prey.

The vast majority of dredging considered under this Opinion will be limited to the navigation channel in sturgeon rivers with intermittent maintenance dredging occurring along the shoreline for maintenance of ports, berths, marinas, and around existing structures like docks. Dredging in the navigation channel, ports, and berths are expected to occur more frequently than privately owned smaller areas along the shores of the river due to the navigational requirements of these larger areas and the funding provided to maintain these areas. A review of the navigation channel dredging anticipated to be completed in the next 5 years was provided by the USACE. That information indicates dredging intervals range from around 6 months in frequently dredged areas, to many years in less frequently dredged areas. Even in frequently dredged areas, a specific dredge footprint is unlikely to be dredged more frequently than every 6 months, because maintenance dredging is not expected to occur in exactly the same dredge footprint in successive dredging events. This should allow sufficient time for these areas to be predominately or completely recolonized, based on the rates of expected recolonization of sturgeon prey species discussed in Section 3.1.7 of this Opinion. Thus, we anticipate that impacts to this feature will be temporary and therefore insignificant, as soft substrate will remain available following maintenance dredging, with only a temporary reduction of prey for foraging. Placement of dredge material in spawning rivers is not an activity covered under this Opinion (e.g., nearshore placement and side-cast dredging) and therefore adverse affects from direct burial of prey resources caused by these activities will not occur.

While the areas adjacent to dredging will not be removed, they may receive sedimentation from the turbidity generated during dredging, especially from agitation dredging such as water-injection dredging. There is evidence that benthic communities in areas immediately adjacent to dredging may benefit from the release of organic materials caused by dredging (Newell et al. 1998). (Newell et al. 1998) report several studies documented enhanced species diversity and population density of organisms in areas immediately adjacent to dredge sites due to these releases (Biggs 1968; Ingle 1952; Jones and Candy 1981; Oviatt et al. 1981; Poiner and Kennedy 1984; Sherk Jr. 1972; Stephenson et al. 1978; Walker and O'Donnell 1981). Because we anticipate dredged areas will have an opportunity to recover, and areas immediately adjacent to dredging areas will not be removed and (and possibly enhanced), we believe foraging resources used by sturgeon will continue be able to fulfill its ecological role/function.

3.3.2.3 PBF 3. Unobstructed Water of Appropriate Depth

We believe that the activities covered under this Opinion will have an insignificant effect on PBF 3. The Sturgeon PDCs limit the upstream distance that work can occur under this Opinion to protect spawning habitat. Spawning can only be successful if adult Atlantic sturgeon are able to safely and efficiently move from downstream areas into upstream spawning habitats. In addition, larvae and juvenile Atlantic sturgeon must be able to safely and efficiently travel from the upstream spawning areas downstream to nursery and foraging habitat. Obstructions can be caused by: locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc. Similarly, water depth is also very important. Minimum water depths for Atlantic sturgeon are necessary to: (1) allow adult fish to access spawning substrate, (2) adequately hydrate and aerate newly deposited eggs, and (3) facilitate successful development and downstream movement of newly spawned Atlantic sturgeon. To accommodate the body depth and spawning behavior of adult Atlantic sturgeon, water depths no less than 1.2 m (4 ft) are likely necessary when animals are present. Together, these characteristics support: the unimpeded movement of adults to and from spawning sites; seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; as well as staging, resting, or holding of subadults or spawning condition adults.

Dredging associated with this Opinion could cause obstructions via turbidity plumes, equipment noise, and the presence of equipment associated with activities covered under this Opinion. However, as described in Section 3.1.1 of this Opinion, the information currently available suggests sturgeon showed no behavioral response directly correlated to these stimuli (i.e., dredge noise, dredge equipment, or resultant turbidity). The level of noise associated with projects covered under this Opinion is limited to levels that would not result in harm to the species and is not expected to cause behavioral changes, as discussed in Section 3.1.8 of this Opinion. Also, the turbidity generated is expected to have an insignificant effect to sturgeon, as discussed in Section 3.1.1.2 of this Opinion, and would therefore not block access to sturgeon using these rivers. Therefore, we do not anticipate activities covered under this Opinion will deter or obstruct sturgeon from using spawning rivers, as required by PBF 3. The Sturgeon PDCs also state that dredging operations, including related equipment, will not block more than 50% of the river width to allow continued unobstructed migration of sturgeon in spawning rivers (Sturgeon PDCs, Section 2 in Appendix E). Atlantic sturgeon critical habitat occurs within rivers. In rivers, the currents compress the turbidity plume as it moves downstream and settles, reducing the overall area/volume affected by it, as discussed in Section 3.1.1 during the analysis of water quality changes in sturgeon spawning rivers. Thus, the likelihood of turbidity anticipated in association with the proposed action obstructing sturgeon's access to portions of the river decreases with the distance from the dredge. Dredging in Atlantic sturgeon critical habitat will be limited to the channel that is being maintained or ports and berths along the side of the river. Under the Sturgeon PDCs, at least 50% of the channel/river will remain open for sturgeon to traverse without exposure to the dredge equipment or the turbidity plume. Additionally, any limited obstruction that may occur would only last for the duration of the project at issue and therefore be temporary. Thus, turbidity caused by dredging is not anticipated to cause an obstruction or otherwise keep from the PBF from fulfilling its function.

The dredging covered under this Opinion will maintain existing water depths. For dredging following the PDCs of this Opinion, we do not anticipate any changes in water depth that will impede this PBF from fulfilling its function. In-water placement of dredged materials in rivers is not covered under this Opinion as the Sturgeon PDCs do not allow side-cast dredging or other forms of in-water placement that could reduce the available water column depth sturgeon use for migration within the spawning rivers, thereby allowing unobstructed water depths.

3.3.2.4 PBF 4. Water quality conditions

We believe the activities covered under this Opinion will have an insignificant effect on PBF 4. Water quality conditions, especially in the bottom meter of the water column, need to remain with a temperature and DO range that supports spawning; survival of all life stages year after year; and promotes larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and DO values will vary interdependently, and depending on salinity in a particular habitat. For example, 6.0 mg/L DO or greater likely supports juvenile rearing habitat, whereas DO less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25°C. In temperatures greater than 26°C, DO greater than

4.3 mg/L is needed to protect survival and growth. Temperatures of 13 to 26 °C likely support spawning habitat.

As discussed in Section 3.1.1.2 of this Opinion, we believe certain types of dredging associated with the proposed action could reduce the DO concentrations. However, we anticipate those effects will be temporary, and therefore insignificant. The information reported in Section 3.1.1.2 indicates the changes in DO concentrations caused by these forms of dredging are confined to areas relatively close to where the equipment operates and DO concentrations return to background levels within hours. The dredging activities considered under this Opinion are not anticipated to have any measurable effect on water temperature and will not effect this portion of the PBF.

3.3.3 Green, Hawksbill, and Leatherback Sea Turtle Critical Habitat

The action area of this Opinion encompasses all areas designated as green, hawksbill, and leatherback sea turtle critical habitats, which are located in in the U.S. Caribbean, specifically around Culebra Island, Puerto Rico, for green sea turtle critical habitat; Mona Island, Puerto Rico, for hawksbill sea turtle critical habitat; and Sandy Point, St. Croix, U.S. Virgin Islands for leatherback sea turtle critical habitat. The features of each of these critical habitats are provided in Table 24 below.

Table 24. Green, Hawksbill, and Leatherback Sea Turtle Critical Habitat PBFs

Green sea turtle (63 FR 46693, Publication Date: September 2,1998)	Critical habitat for the green sea turtle is designated in the waters surrounding the island of Culebra, Puerto Rico, from the mean high water line seaward to 3 nautical miles. These waters include Culebra's outlying Keys, including Cayo Norte, Cayo Ballena, Cayos Geniquı´, Isla Culebrita, Arrecife Culebrita, Cayo de Luis Pena, Las Hermanas, El Mono, Cayo Lobo, Cayo Lobito, Cayo Botijuela, Alcarraza, Los Gemelos, and Piedra Steven. At that time, essential features to critical habitat were not precisely defined; however, the critical habitat was designated to provide protection mainly for important developmental and resting habitats. Juvenile and adult green sea turtles depend on seagrasses as the principal dietary component of for foraging. In addition, coral reefs and other topographic features within the waters around Culebra Island and surrounding islands and cays provide green turtles with shelter during interforaging periods that serve as refuge from predators.
Hawksbill sea turtles (63 FR 46693, Publication Date: September 2, 1998)	Critical habitat for the hawksbill sea turtle has been designated in the waters surrounding the islands of Mona and Monito, Puerto Rico, from the mean high water line seaward to 3 nautical miles. At that time, essential features to critical habitat were not precisely defined; however, the critical habitat was designated to provide protection mainly for important developmental and resting habitats. Hawksbill sea turtles depend on sponges as their principal dietary component and healthy coral reefs for foraging and shelter habitats.
Leatherback sea turtles (44 FR 8491, Publication Date: March 23, 1979)	Critical habitat for the leatherback sea turtle has been designated in the waters adjacent to Sandy Point on the southwest corner of St. Croix, U.S. Virgin Islands, in waters from the 100-fathom curve shoreward to the level of mean high tide, with boundaries at 17°42′12" N and 64°50′00″W. At that time, essential features to critical habitat were not precisely defined; however, critical habitat for leatherback sea turtles was designated to provide protection to sea turtles using these waters for courting, breeding, and as access to and from nesting areas on Sandy Point Beach, St. Croix, U.S. Virgin Islands.

We believe that there will be no effect from any of the activities analyzed in this Opinion to any of the features of green, hawksbill, or leatherback sea turtle critical habitat because beach nourishment projects are not covered under this Opinion in the U.S. Caribbean where these critical habitat units occur.

3.3.4 Loggerhead Sea Turtle Critical Habitat

Loggerhead sea turtle critical habitat was designated along the southeast Atlantic coast of the United States, around the Florida peninsula, and through the Gulf of Mexico to Texas (79 FR 39855, Publication Date July 10, 2014), of which the portions off of North Carolina, South Carolina, Georgia, and the east coast of Florida and the Florida Keys are within the action area of this Opinion. Loggerhead critical habitat is divided into 5 different units: nearshore reproductive habitat, winter habitat, breeding areas, constricted migratory habitat, and *Sargassum* habitat. All of the features of these habitat units are provided in Table 25. For loggerhead sea turtle critical habitat, additional information about the PBF was provided to explain the PBF, referred to in Table 25 as the PCE.

Table 25. NWA DPS of Loggerhead Sea Turtle Critical Habitat PBFs

Habitat Type	Units	State	PBF	PCE
Nearshore Reproductive Habitat	LOGG-N-3 through N-6 LOGG-N-7 through N-11 LOGG-N-12 and N-13 LOGG-N-14 through N-32 LOGG-N-33 LOGG-N-34 through N-36	North Carolina South Carolina Georgia Florida Florida, Alabama Alabama, Mississippi	Portion of nearshore waters adjacent to nesting beaches that hatchlings use as egress to the open-water environment. Also used by nesting females to transit between beach and open water during the nesting season.	1. Nearshore waters with direct proximity to nesting beaches that support critical aggregations of nesting turtles (e.g., highest density nesting beaches) to 1.6 km (1 mile) offshore **2.** Waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water **3.** Waters with minimal man-made structures that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents
Winter Habitat	LOGG-N-1 and N-2	North Carolina	Warm water habitat south of Cape Hatteras, near the western edge of the Gulf Stream, which supports meaningful aggregations of juveniles and adults	1. Water temperatures above 10°C during the colder months of November through April 2. Continental shelf waters in proximity to the western boundary of the Gulf Stream 3. Water depths between 20-100 m)
Concentrated Breeding Habitat	LOGG-N-17 and N-19	Florida	Sites that support meaningful aggregations of both male and female adult individuals	1. Meaningful concentrations of reproductive male and female loggerheads 2. Proximity to primary Florida migratory corridor 3. Proximity to Florida nesting grounds
Constricted Migratory Corridor Habitat	LOGG-N-1 LOGG-N-17 through N-19	North Carolina Florida	High-use migratory corridors that are constricted (limited in width) by land on one side and the edge of the continental shelf and Gulf Stream on the other side	1. Constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways 2. Passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas

3.3.4.1 Nearshore reproductive habitat

Nearshore reproductive habitat is located within 1 mile from shore in areas with sea turtle nesting beaches and found within the action area from North Carolina to south Florida. We believe that dredging or the placement of materials and the transportation of materials may affect, but is not likely to adversely affect the waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water feature of loggerhead sea turtle critical habitat. We believe the effects to this feature will be insignificant. The General PDCs in Section 1.2 in Appendix B describe beach nourishment covered under this Opinion and the General PDCs in Section 2.2 in Appendix B provide conditions that limit how and where material is placed and minimize lighting on construction equipment. Based on the

PDCs, lighting on construction equipment near nesting beaches will be turtle friendly so as not to disorient hatchlings returning to the ocean. The PDCs also state that material placement and equipment will be staged in a manner that would not block access of ESA-listed species, including the access of nesting sea turtles to the beach or of hatchlings returning to the water. This includes the placement of sand in a manner that would not mound or block access to sea turtle nesting beaches, except for the temporary placement of sand berms during beach nourishment projects designed to minimize turbidity during placement of sand. In addition, beach placement covered under this Opinion, if any, will be limited to activities that follow all PDCs in this Opinion such as those designed to ensure project activities and equipment do not obstruct species movement such as that of sea turtles entering or exiting the beach when nesting or species moving along the shoreline (General PDC Section 2.2 in Appendix B).

Nearshore reproductive habitat only extends 1 mile from shore and most borrow sites and all ODMDS sites occur outside of this range. A few limited borrow sites associated with entrance channel dredging or a specific beach nourishment project occur within this range, but would only be used when work in the associated channel or beach is completed. Since projects during nesting season would require additional review and permitting outside of NMFS purview, we anticipate the dredging or placement of materials in these areas will be limited. If these areas are used during nesting season, activities will need to follow the same in-water PDCs discussed above for beach nourishment projects designed to ensure placement of material does not obstruct species movement such as that of sea turtles entering or exiting the beach when nesting or species moving along the shoreline.

3.3.4.2 Winter habitat

Winter habitat is restricted to waters off of North Carolina. Dredging or material placement in this location would be limited to ODMDS or borrow sites due to its distance of over 4 miles from shore. We believe that dredging or material placement, transportation of materials, or geophysical surveys would have no effect on the features of loggerhead sea turtle critical habitat, specifically the water temperature, proximity to the Gulf Stream, or availability of water depths between 20-100m. Even minor changes in depth from dredging, placement, or geophysical surveys would not change the depth range defined in the critical habitat feature.

3.3.4.3 Concentrated Breeding Habitat

Breeding areas within the action area are limited to 2 sections off the central and south Florida coast. We believe that placement of materials, transportation or materials, or geotechnical surveys will have no effect on the features of loggerhead sea turtle critical habitat breeding areas due to the PDCs restricting placement of equipment and materials that block access to nesting beaches.

The features are related to density of reproductive loggerheads and their proximity to the migratory corridors and nesting grounds. None of these activities will adversely affect the density of turtles or change migratory pathways.

Dredging of materials by hopper dredging or relocation of turtles may affect, but is not likely to adversely affect the high densities of reproductive male and female loggerheads feature of loggerhead sea turtle critical habitat breeding areas. We believe that a potential minor changes in numbers from hopper dredge take or temporarily relocating of turtles (adverse effects to the species) during discrete projects would have a temporary and therefore insignificant effect on the "high density" feature. Relocation trawling temporarily relocates turtles approximately 3 miles away, and therefore within the same general breeding area. Relocation accordingly does not significantly change the density of reproductive loggerheads in the area. The potential take of a small number of turtles by dredging in a limited area would not meaningfully change the high density feature.

3.3.4.4 Constricted migratory habitat

Constricted migratory habitat within the action area is located along the portion of the North Carolina shoreline and from central Florida to the Keys. We believe that dredging, placement of materials, transportation or materials, or geotechnical surveys will have no effect on the features of loggerhead sea turtle critical habitat constricted migratory areas. The features of this habitat are related to the geographic features of the area allowing passage along the continental shelf and migration to and from nesting, breeding, and foraging areas. None of these maintenance activities are large enough in scope or scale to affect the migratory pathways and would therefore have no effect to this feature.

3.3.4.5 Sargassum habitat

Sargassum habitat is located offshore in the action area along the coast from North Carolina to the Florida Keys. The critical habitat area nears shore along a portion of the Florida coast and includes 2 identified ODMDS sites. We believe that dredging, placement of materials, transportation of materials, or geotechnical surveys will have no effect on the features of loggerhead sea turtle critical habitat *Sargassum* habitat. The features of this habitat are the currents and temperatures that support floating communities accumulating floating mats of *Sargassum* and associated flora and fauna. These mats are used by loggerhead sea turtle hatchlings for foraging and refuge. None of the activities in this Opinion are large enough in scope or scale to damage these floating mats. Transportation through the area may temporarily separate these mats, but will not remove, destroy, or pollute them allowing them to continue to function for the purposes of hatching development.

3.3.5 Johnson's Seagrass Critical Habitat

The action area of this Opinion encompasses all areas designated as Johnson's seagrass critical habitat, which is located in 10 separate areas in the coastal lagoon system of southeast Florida. The Johnson's seagrass critical habitat PBFs are provided in Table 26 below.

Table 26. Johnson's Seagrass Critical Habitat PBFs

Johnson's seagrass (65 FR 17786, Publication Date: April 5, 2000)	Based on the best available information, general physical and biological features of the critical habitat areas include adequate water quality, salinity levels, water transparency, and stable, unconsolidated sediments that are free from physical disturbance.

As discussed in Section 1.1.1 of the Johnson's Seagrass PDCs in Appendix D, the following areas are not considered critical habitat (items 1-3 below) or are areas that we believe are not functioning as critical habitat (items 4-7 below) and activities occurring in these areas will have no effect on critical habitat. It is important to note that areas not considered critical habitat (items 1-3 below) may still have effects to the species if it occurs in this area, as was discussed in Section 3.2.1 of this Opinion.

1. Areas within the geographic boundary of critical habitat that do not support all 4 physical and biological features listed above. This is the only designated critical habitat within the SARBO action area that requires the presence of all of the physical and biological features to be functioning critical habitat.
    a. Areas deeper than 13 ft are assumed to be too deep to support adequate water transparency for Johnson's seagrass growth. Studies show that Johnson's seagrass occurs in waters shallower than 10-13 ft (3-4 m) (NMFS 2007a). Water depths greater than 13 ft are not believed to provide the water transparency necessary for sufficient sunlight to reach the sea floor to support Johnson's seagrass growth. Therefore, we consider the water transparency PBF of Johnson's seagrass critical habitat not to be present in waters deeper than 13 ft. All areas within the critical habitat designation, but in waters deeper than 13 ft, are accordingly not considered to be functioning critical habitat for the analysis in 2020 SARBO.
    b. Consolidated sediments (e.g., hardbottom).
    c. Areas with accumulated material that precludes seagrass growth (e.g., thick muck).
2. The 18.5–km-long portion of the navigation channel within the Intracoastal Waterway that occurs within the geographical limits of Johnson's seagrass is excluded from critical habitat (65 FR 17786, Publication Date April, 5, 2000, page 17791, item 4).
3. A portion of northern Biscayne Bay. The geographic limits of Johnson's seagrass critical habitat align with the boundaries of the Biscayne Bay Aquatic Preserve. The USACE and NMFS have interpreted this to mean that some of the man-made canals in this area are not in Johnson's seagrass critical habitat, because they are not part of the Biscayne Bay Aquatic Preserve. A GIS layer is available as a tool to clarify whether a particular project is located inside or outside of critical habitat based on the Aquatic Preserve geographic boundary (https://sero.nmfs.noaa.gov/maps_gis_data/index.html).

We believe activities covered under this Opinion will have either no effect or an insignificant effect on Johnson's seagrass critical habitat.

- <u>Maintenance dredging within the IWW:</u> Approximately 11.5 miles (~18.5 km) of the IWW run through Johnson's seagrass critical habitat; however, this federal navigation channel was excluded from designated Johnson's seagrass critical habitat (65 FR 17786, Publication Date April 5, 2000) and therefore activities in the IWW channel will have no effect to Johnson's seagrass critical habitat. Any Johnson's seagrass growing within the channel is addressed separately in the effects to the species analysis in this Opinion. Effects from turbidity and sedimentation to adjacent areas of critical habitat are considered separately, below.

- Borrow sites: Dredging in borrow areas in Johnson's seagrass critical habitat is limited to only the borrow site located at the entrance of Jupiter Inlet where the inlet meets the IWW, which creates a shoal that is frequently dredged for sand used for beach nourishment projects. This 20.56 acre borrow site referred to as Baker's Haulover is in a dynamic shoal area that lacks seagrasses and is frequently dredged and therefore does not support the "stable, unconsolidated sediments that are free from physical disturbance" feature of critical habitat and is therefore not functioning as critical habitat.
- Geotechnical surveys: Geotechnical surveys are prohibited in areas where Johnson's seagrass is present and where the features that support Johnson's seagrass are present (defined in Johnson's Seagrass PDC Section 1.2) and therefore there will be no effect to critical habitat from this activity.
- In-water placement: The Johnson's seagrass PDCs do not cover projects that would have in-water placement of dredged materials in Johnson's seagrass critical habitat and therefore there will be no effect to critical habitat from this activity.
- Dredging and in-water placement deeper than 13 ft: The Johnson's Seagrass PDCs (Johnson's seagrass PDC Section 3 in Appendix D) limit material placement within the range of Johnson's seagrass to only areas that are deeper than 13 ft and therefore not expected to support the PBFs of Johnson's seagrass. Studies show that Johnson's seagrass occurs in waters shallower than 10-13 ft (3-4 m) (NMFS 2007a). Areas deeper than 13 ft are assumed to be too deep to support adequate water transparency for Johnson's seagrass growth, which is a PBF of critical habitat and if it is not present then the area is not functioning as Johnson's seagrass critical habitat. Because these areas are not functioning critical habitat, dredging or material placement in these deeper water areas will have no effect to Johnson's seagrass critical habitat.
- Placement of equipment: We believe that the placement of equipment and the transportation of materials by pipeline, hopper dredge, barge, or scowl, may temporarily affect the PBFs of water quality, transparency and stable unconsolidated sediments. These activities would not be expected to impact salinity. Placement of equipment in Johnson's seagrass critical habitat may result in turbidity, decreased water quality, and destabilization of sediments. However, we expect any effects will be insignificant. Placement of equipment in Johnson's seagrass critical habitat would only temporarily impact its availability to the species, and the habitat would return to being fully functional following removal of the equipment. Additionally, the Johnson's seagrass PDCs in Section 2 in Appendix D state that placement of equipment and materials will avoid areas with any seagrasses, which we would expect to also be areas with functioning critical habitat. In cases where pipeline placement cannot avoid seagrass, floating pipelines will be used instead of anchored pipelines. We expect any effects associated with equipment placement in functioning critical habitat to be insignificant as they will be temporary and limited in geographic extent, and the critical habitat would remain fully functional upon removal.
- Turbidity and Sedimentation: Dredging and material placement may temporarily affect the water transparency PBF in the areas surrounding these activities, such as the areas adjacent to maintenance dredging in the IWW, or other dredging near the boundary of critical habitat. The Johnson's seagrass PDCs (Johnson's seagrass PDCs in Section 2 in Appendix D) limit the type of dredging equipment that can be used within the range of Johnson's seagrass, to

minimize turbidity generated that could affect the adequate water quality and water transparency PBF in surrounding areas and to minimize that turbidity from resulting in sedimentation that could affect the stable, unconsolidated sediments that are free from physical disturbance PBF in areas surrounding dredging, as discussed above. We believe that these minimization measures will ensure that turbidity and sedimentation generated during dredging covered under this Opinion will have an insignificant effect on critical habitat in the areas surrounding dredging.

3.3.6 North Atlantic Right Whale Critical Habitat

North Atlantic right whale critical habitat was designated in 2 areas along the Atlantic coast of the United States, of which the entire portion of the Southeast U.S. Calving Area (Unit 2) occurs within the action the action area of this Opinion. The North Atlantic right whale critical habitat PBFs are provided in Table 27 below.

Table 27. North Atlantic Right Whale Critical Habitat PBFs

North Atlantic right whale (81 FR 4837, Publication Date: January 27, 2016)	Critical habitat includes 2 areas (Units) located in the Gulf of Maine and Georges Bank Region (Unit 1) and off the coast of North Carolina, South Carolina, Georgia and Florida (Unit 2). Unit 2: The physical features essential to the conservation of the North Atlantic right whale, which provide calving area functions in Unit 2, are: • Sea surface conditions associated with Force 4 or less on the Beaufort Scale • Sea surface temperatures of 7°C to 17°C • Water depths of 6 to 28 m, where these features simultaneously co-occur over contiguous areas of at least 231 nmi² of ocean waters during the months of November through April. When these features are available, they are selected by right whale cows and calves in dynamic combinations that are suitable for calving, nursing, and rearing, and which vary, within the ranges specified, depending on factors such as weather and age of the calves.

We believe that there will be no effect to North Atlantic right whale critical habitat from any of the activities analyzed under this Opinion. The features of North Atlantic right whale critical habitat were designated to provide calving areas, which include specific sea surface conditions, sea surface temperatures, and water depth needed to be available for calving, nursing, and rearing North Atlantic right whale calves. Maintenance dredging, transportation of dredged materials, material placement, or dredging surveys allowed under this Opinion will have no effect on the sea state or temperature and will not change the availability of waters 20-92 ft deep, as defined to be the depth needed in the critical habitat rule. Specifically, dredging must maintain previously existing depths and open water material placement in areas where whales may co-occur will not change the water depth feature in a way that is perceptible to whales.

4 Status of Species Likely to be Adversely Affected

4.1 Status of Species Likely to be Adversely Affected

4.1.1 Sea Turtles

There are 5 sea turtles (green [NA and SA DPSs], hawksbill, Kemp's ridley, leatherback, and loggerhead) that travel widely throughout the South Atlantic, Gulf of Mexico and the Caribbean, and may be adversely affected by the proposed action. Section 4.1.1.1 of this Opinion will address the general threats that confront all sea turtle species. The remainder of Section 4.1.1 (Sections 4.1.1.2 –4.1.1.5) will address information on the distribution, life history, population structure, abundance, population trends, and unique threats to each species of sea turtle.

4.1.1.1 General Threats Faced by All Sea Turtle Species

Sea turtles face numerous natural and man-made threats that shape their status and affect their ability to recover. Many of the threats are either the same or similar in nature for all listed sea turtle species, those identified in this section are discussed in a general sense for all sea turtles. Threat information specific to a particular species are then discussed in the corresponding status sections where appropriate.

4.1.1.1.1 Fisheries

Incidental bycatch in commercial fisheries is identified as a major contributor to past declines, and threat to future recovery, for all of the sea turtle species (NMFS et al. 2011; NMFS and USFWS 1991; NMFS and USFWS 1992; NMFS and USFWS 1993; NMFS and USFWS 2008). Domestic fisheries often capture, injure, and kill sea turtles at various life stages. Sea turtles in the pelagic environment are exposed to U.S. Atlantic pelagic longline fisheries. Sea turtles in the benthic environment in waters off the coastal United States are exposed to a suite of other fisheries in federal and state waters. These fishing methods include trawls, gillnets, purse seines, hook-and-line gear (including bottom longlines and vertical lines [e.g., bandit gear, handlines, and rod-reel]), pound nets, and trap fisheries. Refer to the Environmental Baseline section of this Opinion for more specific information regarding federal and state managed fisheries affecting sea turtles within the action area). The Southeast U.S. shrimp fisheries have historically been the largest fishery threat to benthic sea turtles in the southeastern United States, and continue to interact with and kill large numbers of sea turtles each year.

In addition to domestic fisheries, sea turtles are subject to direct as well as incidental capture in numerous foreign fisheries, further impeding the ability of sea turtles to survive and recover on a global scale. For example, pelagic stage sea turtles, especially loggerheads and leatherbacks, circumnavigating the Atlantic are susceptible to international longline fisheries including the Azorean, Spanish, and various other fleets (Aguilar et al. 1995; Bolten et al. 1994). Bottom longlines and gillnet fishing is known to occur in many foreign waters, including (but not limited to) the northwest Atlantic, western Mediterranean, South America, West Africa, Central America, and the Caribbean. Shrimp trawl fisheries are also occurring off the shores of numerous foreign countries and pose a significant threat to sea turtles similar to the impacts seen in U.S. waters. Many unreported takes or incomplete records by foreign fleets make it difficult to characterize the total impact that international fishing pressure is having on listed sea turtles. Nevertheless, international fisheries represent a continuing threat to sea turtle survival and recovery throughout their respective ranges.

4.1.1.1.2 Non-Fishery In-Water Activities

There are also many non-fishery impacts affecting the status of sea turtle species, both in the ocean and on land. In nearshore waters of the United States, the construction and maintenance of federal navigation channels has been identified as a source of sea turtle mortality. Hopper dredges, which are frequently used in ocean bar channels and sometimes in harbor channels and offshore borrow areas, move relatively rapidly and can entrain and kill sea turtles (NMFS 1997). Sea turtles entering coastal or inshore areas have also been affected by entrainment in the cooling-water systems of electrical generating plants. Other nearshore threats include harassment and/or injury resulting from private and commercial vessel operations, military detonations and training exercises, in-water construction activities, and scientific research activities.

4.1.1.1.3 Coastal Development and Erosion Control

Coastal development can deter or interfere with nesting, affect nesting success, and degrade nesting habitats for sea turtles. Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and nourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). These factors may decrease the amount of nesting area available to females and change the natural behaviors of both adults and hatchlings, directly or indirectly, through loss of beach habitat or changing thermal profiles and increasing erosion, respectively (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). In addition, coastal development is usually accompanied by artificial lighting which can alter the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings that are drawn away from the water (Witherington and Bjorndal 1991). In-water erosion control structures such as breakwaters, groins, and jetties can impact nesting females and hatchlings as they approach and leave the surf zone or head out to sea by creating physical blockage, concentrating predators, creating longshore currents, and disrupting of wave patterns.

4.1.1.1.4 Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., dichlorodiphenyltrichloroethane [DDT], polychlorinated biphenyls [PCB], and perfluorinated chemicals), and others that may cause adverse health effects to sea turtles (Garrett 2004; Grant and Ross 2002; Hartwell 2004; Iwata et al. 1993). Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges may directly injure individuals through skin contact with oils (Geraci 1990), inhalation at the water's surface and ingesting compounds while feeding (Matkin and Saulitis 1997). Hydrocarbons also have the potential to impact prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area.

The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. An assessment has been completed on the injury to Gulf of Mexico marine life, including sea turtles, resulting from the spill (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. The spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. Information on the spill impacts to individual sea turtle species is presented in the Status of the Species sections for each species.

Marine debris is a continuing problem for sea turtles. Sea turtles living in the pelagic environment commonly eat or become entangled in marine debris (e.g., tar balls, plastic bags/pellets, balloons, and ghost fishing gear) as they feed along oceanographic fronts where debris and their natural food items converge. This is especially problematic for sea turtles that spend all or significant portions of their life cycle in the pelagic environment (i.e., leatherbacks, juvenile loggerheads, and juvenile green turtles).

4.1.1.1.5 Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change, exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see http://www.climate.gov).

Climate change impacts on sea turtles currently cannot be predicted with any degree of certainty; however, significant impacts to the hatchling sex ratios of sea turtles may result (NMFS and USFWS 2007b). In sea turtles, sex is determined by the ambient sand temperature (during the middle third of incubation) with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25°-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007b).

The effects from increased temperatures may be intensified on developed nesting beaches where shoreline armoring and construction have denuded vegetation. Erosion control structures could potentially result in the permanent loss of nesting beach habitat or deter nesting females (National Research Council 1990b). These impacts will be exacerbated by sea level rise. If females nest on the seaward side of the erosion control structures, nests may be exposed to repeated tidal overwash (NMFS and USFWS 2007b). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005). The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

Other changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, DO levels, nutrient distribution, etc.) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish, etc.) which could ultimately affect the primary foraging areas of sea turtles.

4.1.1.1.6 Other Threats

Predation by various land predators is a threat to developing nests and emerging hatchlings. The major natural predators of sea turtle nests are mammals, including raccoons, dogs, pigs, skunks, and badgers. Emergent hatchlings are preyed upon by these mammals as well as ghost crabs, laughing gulls, and the exotic South American fire ant (*Solenopsis invicta*). In addition to natural predation, direct harvest of eggs and adults from beaches in foreign countries continues to be a problem for various sea turtle species throughout their ranges (NMFS and USFWS 2008).

Diseases, toxic blooms from algae and other microorganisms, and cold stunning events are additional sources of mortality that can range from local and limited to wide-scale and impacting hundreds or thousands of animals.

4.1.1.2 Green Sea Turtle (Information Relevant to All DPSs)

The green sea turtle was originally listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered. On April 6, 2016, the original listing was replaced with the listing of 11 DPSs (81 FR 20057, Publication Date April 6, 2016) (Figure 24). The Mediterranean, Central West Pacific, and Central South Pacific DPSs were listed as endangered. The North Atlantic, South Atlantic, Southwest Indian, North Indian, East Indian-West Pacific, Southwest Pacific, Central North Pacific, and East Pacific DPSs were listed as threatened. For the purposes of this consultation, only the SA DPS and NA DPS will be considered, as they are the only 2 DPSs with individuals occurring in the Atlantic and Gulf of Mexico waters of the United States.

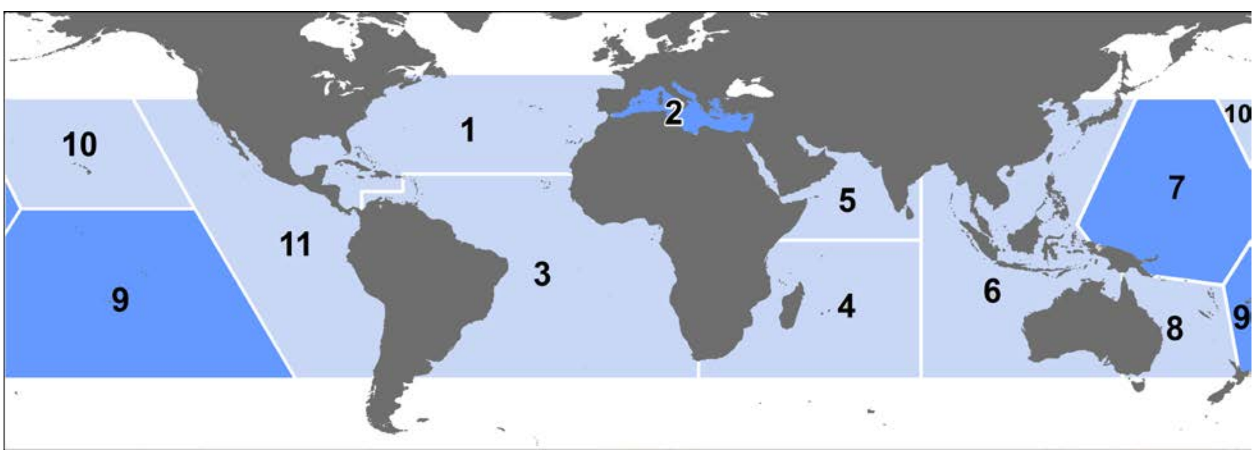

Figure 24. Threatened (light) and endangered (dark) green turtle DPSs: 1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11. East Pacific.

4.1.1.2.1 Species Description and Distribution

The green sea turtle is the largest of the hardshell marine turtles, growing to a weight of 350 lb (159 kg [kilogram]) with a straight carapace length of greater than 3.3 ft (1 m). Green sea turtles have a smooth carapace with 4 pairs of lateral (or costal) scutes and a single pair of elongated prefrontal scales between the eyes. They typically have a black dorsal surface and a white ventral surface, although the carapace of green sea turtles in the Atlantic Ocean has been known to change in color from solid black to a variety of shades of grey, green, or brown and black in starburst or irregular patterns (Lagueux 2001).

With the exception of post-hatchlings, green sea turtles live in nearshore tropical and subtropical waters where they generally feed on marine algae and seagrasses. They have specific foraging grounds and may make large migrations between these forage sites and natal beaches for nesting (Hays et al. 2001). Green sea turtles nest on sandy beaches of mainland shores, barrier islands, coral islands, and volcanic islands in more than 80 countries worldwide (Hirth 1997). The 2 largest nesting populations are found at Tortuguero, on the Caribbean coast of Costa Rica (part of the NA DPS), and Raine Island, on the Pacific coast of Australia along the Great Barrier Reef.

Differences in mitochondrial deoxyribonucleic acid (DNA) properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; FitzSimmons et al. 2006). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. Within U.S. waters individuals from both the NA and SA DPSs can be found on foraging grounds. While there are currently no in-depth studies available to determine the percent of NA and SA DPS individuals in any given location, 2 small-scale studies provide an insight into the degree of mixing on the foraging grounds. An analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of Mexico) found approximately 4% of individuals came from nesting stocks in the SA DPS (specifically Suriname, Aves Island, Brazil, Ascension Island, and Guinea Bissau) (Foley et al. 2007). On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the Aves Island/Suriname nesting assemblage, which is part of the SA DPS (Bass and Witzell 2000). All of the individuals in both studies were benthic juveniles. Available information on green turtle migratory behavior indicates that long distance dispersal is only seen for juvenile turtles. This suggests that larger adult-sized turtles return to forage within the region of their natal rookeries, thereby limiting the potential for gene flow across larger scales (Monzón-Argüello et al. 2010). While all of the mainland U.S. nesting individuals are part of the NA DPS, the U.S. Caribbean nesting assemblages are split between the NA and SA DPS. Nesters in Puerto Rico are part of the NA DPS, while those in the U.S. Virgin Islands are part of the SA DPS. We do not currently have information on what percent of individuals on the U.S. Caribbean foraging grounds come from which DPS.

North Atlantic DPS Distribution

The NA DPS boundary is illustrated in Figure 24. Four regions support nesting concentrations of particular interest in the NA DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and

Quintana Roo), U.S. (Florida), and Cuba. By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey 2001).

The complete nesting range of NA DPS green sea turtles within the southeastern United States includes sandy beaches between Texas and North Carolina, as well as Puerto Rico (Dow et al. 2007; NMFS and USFWS 1991). The vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Principal U.S. nesting areas for green sea turtles are in eastern Florida, predominantly Brevard south through Broward counties.

In U.S. Atlantic and Gulf of Mexico waters, green sea turtles are distributed throughout inshore and nearshore waters from Texas to Massachusetts. Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984; Hildebrand 1982; Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1991; Wershoven and Wershoven 1992). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatán Peninsula.

SA DPS Distribution

The SA DPS boundary is shown in Figure 24, and includes the U.S. Virgin Islands in the Caribbean. The SA DPS nesting sites can be roughly divided into 4 regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, island nesting sites).

The in-water range of the SA DPS is widespread. In the eastern South Atlantic, significant sea turtle habitats have been identified, including green turtle feeding grounds in Corisco Bay, Equatorial Guinea/Gabon (Formia 1999); Congo; Mussulo Bay, Angola (Carr and Carr 1991); as well as Principe Island. Juvenile and adult green turtles utilize foraging areas throughout the Caribbean areas of the South Atlantic, often resulting in interactions with fisheries occurring in those same waters (Dow et al. 2007). Juvenile green turtles from multiple rookeries also frequently utilize the nearshore waters off Brazil as foraging grounds as evidenced from the frequent captures by fisheries (Lima et al. 2010; López-Barrera et al. 2012; Marcovaldi et al. 2009). Genetic analysis of green turtles on the foraging grounds off Ubatuba and Almofala, Brazil show mixed stocks coming primarily from Ascension, Suriname and Trindade as a secondary source, but also Aves, and even sometimes Costa Rica (North Atlantic DPS)(Naro-Maciel et al. 2007; Naro-Maciel et al. 2012). While no nesting occurs as far south as Uruguay and Argentina, both have important foraging grounds for South Atlantic green turtles (Gonzalez Carman et al. 2011; Lezama 2009; López-Mendilaharsu et al. 2006; Prosdocimi et al. 2012; Zinno 2012).

4.1.1.2.2 Life History Information

Green sea turtles reproduce sexually, and mating occurs in the waters off nesting beaches and along migratory routes. Mature females return to their natal beaches (i.e., the same beaches where they were born) to lay eggs (Balazs 1979; Frazer and Ehrhart 1985) every 2-4 years while males are known to reproduce every year (Balazs 1983). In the southeastern United States, females generally nest between June and September, and peak nesting occurs in June and July (Witherington and Ehrhart 1989b). During the nesting season, females nest at approximately 2-week intervals, laying an average of 3-4 clutches (Johnson and Ehrhart 1996). Clutch size often varies among subpopulations, but mean clutch size is approximately 110-115 eggs. In Florida, green sea turtle nests contain an average of 136 eggs (Witherington and Ehrhart 1989b). Eggs incubate for approximately 2 months before hatching. Hatchling green sea turtles are approximately 2 inches (5 cm) in length and weigh approximately 0.9 ounces (25 grams). Survivorship at any particular nesting site is greatly influenced by the level of man-made stressors, with the more pristine and less disturbed nesting sites (e.g., along the Great Barrier Reef in Australia) showing higher survivorship values than nesting sites known to be highly disturbed (e.g., Nicaragua) (Campell and Lagueux 2005; Chaloupka and Limpus 2005).

After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. This early oceanic phase remains one of the most poorly understood aspects of green sea turtle life history (NMFS and USFWS 2007a). Green sea turtles exhibit particularly slow growth rates of about 0.4-2 inches (1-5 cm) per year (Green 1993), which may be attributed to their largely herbivorous, low-net energy diet (Bjorndal 1982). At approximately 8-10 inches (20-25 cm) carapace length, juveniles leave the pelagic environment and enter nearshore developmental habitats such as protected lagoons and open coastal areas rich in sea grass and marine algae. Growth studies using skeletochronology indicate that green sea turtles in the western Atlantic shift from the oceanic phase to nearshore developmental habitats after approximately 5-6 years (Bresette et al. 2006; Zug and Glor 1998). Within the developmental habitats, juveniles begin the switch to a more herbivorous diet, and by adulthood feed almost exclusively on seagrasses and algae (Ingle and Smith 1974), although some populations are known to also feed heavily on invertebrates (Carballo et al. 2002). Green sea turtles mature slowly, requiring 20-50 years to reach sexual maturity (Chaloupka and Musick 1997; Hirth 1997).

While in coastal habitats, green sea turtles exhibit site fidelity to specific foraging and nesting grounds, and it is clear they are capable of "homing in" on these sites if displaced (McMichael et al. 2003). Reproductive migrations of Florida green sea turtles have been identified through flipper tagging and/or satellite telemetry. Based on these studies, the majority of adult female Florida green sea turtles are believed to reside in nearshore foraging areas throughout the Florida Keys and in the waters southwest of Cape Sable, and some post-nesting turtles also reside in Bahamian waters as well (NMFS and USFWS 2007a).

4.1.1.2.3 Status and Population Dynamics

Accurate population estimates for marine turtles do not exist because of the difficulty in sampling turtles over their geographic ranges and within their marine environments. Nonetheless, researchers have used nesting data to study trends in reproducing sea turtles over time. A summary of nesting trends and nester abundance is provided in the most recent status review for the species (Seminoff et al. 2015), with information for each of the DPSs.

North Atlantic DPS

The NA DPS is the largest of the 11 green turtle DPSs, with an estimated nester abundance of over 167,000 adult females from 73 nesting sites. Overall this DPS is also the most data rich. Eight of the sites have high levels of abundance (i.e., <1000 nesters), located in Costa Rica, Cuba, Mexico, and Florida. All major nesting populations demonstrate long-term increases in abundance (Seminoff et al. 2015).

Tortuguero, Costa Rica is by far the predominant nesting site, accounting for an estimated 79% of nesting for the DPS (Seminoff et al. 2015). Nesting at Tortuguero appears to have been increasing since the 1970's, when monitoring began. For instance, from 1971-1975 there were approximately 41,250 average annual emergences documented and this number increased to an average of 72,200 emergences from 1992-1996 (Bjorndal et al. 1999). Troëng and Rankin (2005) collected nest counts from 1999-2003 and also reported increasing trends in the population consistent with the earlier studies, with nest count data suggesting 17,402-37,290 nesting females per year (NMFS and USFWS 2007a). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica population's growing at 4.9% annually.

In the continental United States, green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida (Meylan et al. 1994; Weishampel et al. 2003). Occasional nesting has also been documented along the Gulf Coast of Florida (Reece et al. 2005). Green sea turtle nesting is documented annually on beaches of North Carolina, South Carolina, and Georgia, though nesting is found in low quantities (up to tens of nests) (nesting databases maintained on www.seaturtle.org).

In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring (Figure 25). According to data collected from Florida's index nesting beach survey from 1989-2017, green sea turtle nest counts across Florida have increased dramatically, from a low of 267 in the early 1990s to a high of 38,954 in 2017. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in 2010 and 2011, and a return to the trend of biennial peaks in abundance thereafter (Figure 25). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9% at that time. Increases have been even more rapid in recent years.

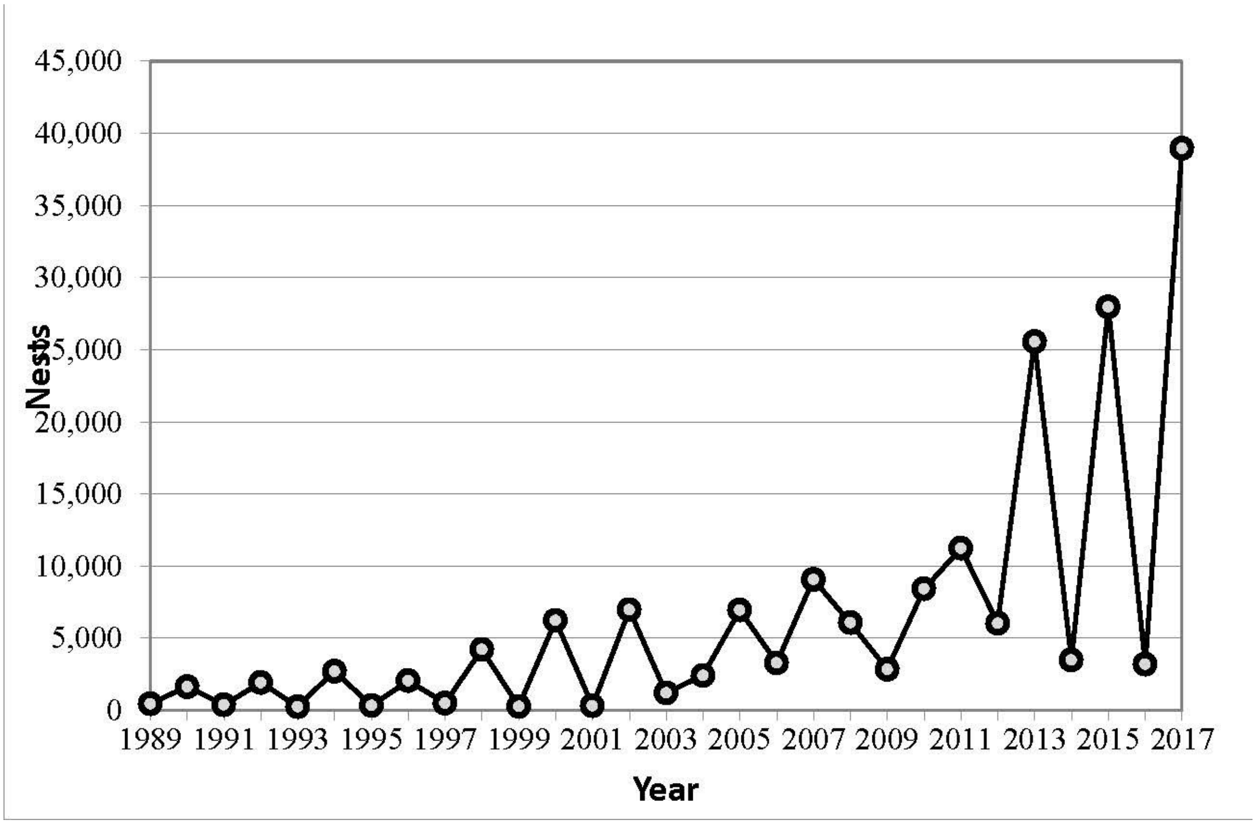

Figure 25. Green sea turtle nesting at Florida index beaches since 1989

Similar to the nesting trend found in Florida, in-water studies in Florida have also recorded increases in green turtle captures at the Indian River Lagoon site, with a 661% increase over 24 years (Ehrhart et al. 2007), and the St Lucie Power Plant site, with a significant increase in the annual rate of capture of immature green turtles (straight carpace length <90 cm) from 1977 to 2002 or 26 years (3,557 green turtles total; M. Bressette, Inwater Research Group, unpublished data; (Witherington et al. 2006).

SA DPS

The SA DPS is large, estimated at over 63,000 nesters, but data availability is poor. More than half of the 51 identified nesting sites (37) did not have sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). This includes some sites, such as beaches in French Guiana, which are suspected to have large numbers of nesters. Therefore, while the estimated number of nesters may be substantially underestimated, we also do not know the population trends at those data-poor beaches. However, while the lack of data was a concern due to increased uncertainty, the overall trend of the SA DPS was not considered to be a major concern as some of the largest nesting beaches such as Ascension Island, Aves Island (Venezuela), and Galibi (Suriname) appear to be increasing. Others such as Trindade (Brazil), Atol das Rocas (Brazil), and Poilão and the rest of Guinea-Bissau seem to be stable or do not have sufficient data to make a determination. Bioko (Equatorial Guinea) appears to be in decline but has less nesting than the other primary sites (Seminoff et al. 2015).

In the U.S., nesting of SA DPS green turtles occurs on the beaches of the U.S. Virgin Islands, primarily on Buck Island. There is insufficient data to determine a trend for Buck Island nesting, and it is a smaller rookery, with approximately 63 total nesters utilizing the beach (Seminoff et al. 2015).

4.1.1.2.4 Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.2.1 of this Opinion.

In addition to general threats, green sea turtles are susceptible to natural mortality from Fibropapillomatosis disease. Fibropapillomatosis disease results in the growth of tumors on soft external tissues (flippers, neck, tail, etc.), the carapace, the eyes, the mouth, and internal organs (gastrointestinal tract, heart, lungs, etc.) of turtles (Aguirre et al. 2002; Herbst 1994; Jacobson et al. 1989). These tumors range in size from 0.04 inches (0.1 cm) to greater than 11.81 inches (30 cm) in diameter and may affect swimming, vision, feeding, and organ function (Aguirre et al. 2002; Herbst 1994; Jacobson et al. 1989). Presently, scientists are unsure of the exact mechanism causing this disease, though it is believed to be related to both an infectious agent, such as a virus (Herbst 1995), and environmental conditions (e.g., habitat degradation, pollution, low wave energy, and shallow water (Foley et al. 2005). Fibropapillomatosis disease is cosmopolitan, but it has been found to affect large numbers of animals in specific areas, including Hawaii and Florida (Herbst 1994; Jacobson 1990; Jacobson et al. 1991).

Cold-stunning is another natural threat to green sea turtles. Although it is not considered a major source of mortality in most cases, as temperatures fall below 46.4°-50°F (degrees Fahrenheit) (8°-10°C [degrees Celsius]) turtles may lose their ability to swim and dive, often floating to the surface. The rate of cooling that precipitates cold-stunning appears to be the primary threat, rather than the water temperature itself (Milton and Lutz 2003). Sea turtles that overwinter in inshore waters are most susceptible to cold-stunning because temperature changes are most rapid in shallow water (Witherington and Ehrhart 1989a). During January 2010, an unusually large cold-stunning event in the southeastern United States resulted in around 4,600 sea turtles, mostly greens, found cold-stunned, and hundreds found dead or dying. A large cold-stunning event occurred in the western Gulf of Mexico in February 2011, resulting in approximately 1,650 green sea turtles found cold-stunned in Texas. Of these, approximately 620 were found dead or died after stranding, while approximately 1,030 turtles were rehabilitated and released. During this same time frame, approximately 340 green sea turtles were found cold-stunned in Mexico, though approximately 300 of those were subsequently rehabilitated and released.

Whereas oil spill impacts are discussed generally for all species in Section 4.2.1 of this Opinion, specific impacts of the Deepwater Horizon oil spill of 2010 (DWH) spill on green sea turtles are considered here. Impacts to green sea turtles occurred to offshore small juveniles only. A total of 154,000 small juvenile greens (36.6% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. A large number of small juveniles were removed from the population, as 57,300 small juvenile greens are estimated to have died as a result of the exposure. A total of 4 nests (580 eggs) were also translocated during response efforts, with 455 hatchlings released (the fate of which is unknown) (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources, which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

While green turtles regularly use the northern Gulf of Mexico, they have a widespread distribution throughout the entire Gulf of Mexico, Caribbean, and Atlantic, and the proportion of the population using the northern Gulf of Mexico at any given time is relatively low. Although it is known that adverse impacts occurred and numbers of animals in the Gulf of Mexico were reduced as a result of the DWH event, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event, as well as the impacts being primarily to smaller juveniles (lower reproductive value than adults and large juveniles), reduces the impact to the overall population. It is unclear what impact these losses may have caused on a population level, but it is not expected to have had a large impact on the population trajectory moving forward. However, recovery of green turtle numbers equivalent to what was lost in the northern Gulf of Mexico as a result of the spill will likely take decades of sustained efforts to reduce the existing threats and enhance survivorship of multiple life stages (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016).

4.1.1.3 Kemp's Ridley Sea Turtle

The Kemp's ridley sea turtle was listed as endangered on December 2, 1970, under the Endangered Species Conservation Act of 1969, a precursor to the ESA. Internationally, the Kemp's ridley is considered the most endangered sea turtle (Groombridge and Wright (compilers) 1982; Turtle Expert Working Group 2000; Zwinenberg 1977).

4.1.1.3.1 Species Description and Distribution

The Kemp's ridley sea turtle is the smallest of all sea turtles. Adults generally weigh less than 100 lb (45 kg) and have a carapace length of around 2.1 ft (65 cm). Adult Kemp's ridley shells are almost as wide as they are long. Coloration changes significantly during development from the grey-black dorsum and plastron of hatchlings, a grey-black dorsum with a yellowish-white plastron as post-pelagic juveniles, and then to the lighter grey-olive carapace and cream-white or yellowish plastron of adults. There are 2 pairs of prefrontal scales on the head, 5 vertebral scutes, usually 5 pairs of costal scutes, and generally 12 pairs of marginal scutes on the carapace. In each bridge adjoining the plastron to the carapace, there are 4 scutes, each of which is perforated by a pore.

Kemp's ridley habitat largely consists of sandy and muddy areas in shallow, nearshore waters less than 120 ft (37 m) deep, although they can also be found in deeper offshore waters. These areas support the primary prey species of the Kemp's ridley sea turtle, which consist of swimming crabs, but may also include fish, jellyfish, and an array of mollusks.

The primary range of Kemp's ridley sea turtles is within the Gulf of Mexico basin, though they also occur in coastal and offshore waters of the U.S. Atlantic Ocean. Juvenile Kemp's ridley sea turtles, possibly carried by oceanic currents, have been recorded as far north as Nova Scotia. Historic records indicate a nesting range from Mustang Island, Texas, in the north to Veracruz, Mexico, in the south. Kemp's ridley sea turtles have recently been nesting along the Atlantic Coast of the United States, with nests recorded from beaches in Florida, Georgia, and the Carolinas. In 2012, the first Kemp's ridley sea turtle nest was recorded in Virginia. The Kemp's ridley nesting population had been exponentially increasing prior to the recent low nesting years, which may indicate that the population had been experiencing a similar increase. Additional nesting data in the coming years will be required to determine what the recent nesting decline means for the population trajectory.

4.1.1.3.2 Life History Information

Kemp's ridley sea turtles share a general life history pattern similar to other sea turtles. Females lay their eggs on coastal beaches where the eggs incubate in sandy nests. After 45-58 days of embryonic development, the hatchlings emerge and swim offshore into deeper, ocean water where they feed and grow until returning at a larger size. Hatchlings generally range from 1.65-1.89 in (42-48 mm) straight carapace length, 1.26-1.73 in (32-44 mm) in width, and 0.3-0.4 lb (15-20 g) in weight. Their return to nearshore coastal habitats typically occurs around 2 years of age (Ogren 1989), although the time spent in the oceanic zone may vary from 1-4 years or perhaps more (Turtle Expert Working Group 2000). Juvenile Kemp's ridley sea turtles use these nearshore coastal habitats from April through November, but they move towards more suitable overwintering habitat in deeper offshore waters (or more southern waters along the Atlantic coast) as water temperature drops.

The average rates of growth may vary by location, but generally fall within 2.2-2.9 $\pm$ 2.4 inches per year (5.5-7.5 $\pm$ 6.2 cm/year) (Schmid and Barichivich 2006; Schmid and Woodhead 2000). Age to sexual maturity ranges greatly from 5-16 years, though NMFS et al. (2011) determined the best estimate of age to maturity for Kemp's ridley sea turtles was 12 years. It is unlikely that most adults grow very much after maturity. While some sea turtles nest annually, the weighted mean remigration rate for Kemp's ridley sea turtles is approximately 2 years. Nesting generally occurs from April to July. Females lay approximately 2.5 nests per season with each nest containing approximately 100 eggs (Márquez M. 1994).

4.1.1.3.3 Population Dynamics

Of the 7 species of sea turtles in the world, the Kemp's ridley has declined to the lowest population level. Most of the population of adult females nest on the beaches of Rancho Nuevo, Mexico (Pritchard 1969). When nesting aggregations at Rancho Nuevo were discovered in 1947, adult female populations were estimated to be in excess of 40,000 individuals (Hildebrand

1963). By the mid-1980s, however, nesting numbers from Rancho Nuevo and adjacent Mexican beaches were below 1,000, with a low of 702 nests in 1985. Yet, nesting steadily increased through the 1990s, and then accelerated during the first decade of the twenty-first century (Figure 26), which indicates the species is recovering.

It is worth noting that when the Bi-National Kemp's Ridley Sea Turtle Population Restoration Project was initiated in 1978, only Rancho Nuevo nests were recorded. In 1988, nesting data from southern beaches at Playa Dos and Barra del Tordo were added. In 1989, data from the northern beaches of Barra Ostionales and Tepehuajes were added, and most recently in 1996, data from La Pesca and Altamira beaches were recorded. Currently, nesting at Rancho Nuevo accounts for just over 81% of all recorded Kemp's ridley nests in Mexico. Following a significant, unexplained 1-year decline in 2010, Kemp's ridley nests in Mexico increased to 21,797 in 2012 (Burchfield 2013). From 2013 through 2014, there was a second significant decline, as only 16,385 and 11,279 nests were recorded, respectively. More recent data, however, indicated an increase in nesting. In 2015 there were 14,006 recorded nests, and in 2016 overall numbers increased to 18,354 recorded nests (Gladys Porter Zoo 2016). There was a record high nesting season in 2017, with 24,570 nests recorded (J. Pena, pers. comm., August 31, 2017), but nesting for 2018 has declined to 17,945 (Gladys Porter Zoo data presentation by J. Pena, 2018). At this time, it is unclear whether the increases and declines in nesting seen over the past decade represents a population oscillating around an equilibrium point or if nesting will decline or increase in the future.

A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (National Park Service data, http://www.nps.gov/pais/naturescience/strp.htm, http://www.nps.gov/pais/naturescience/current-season.htm). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015.

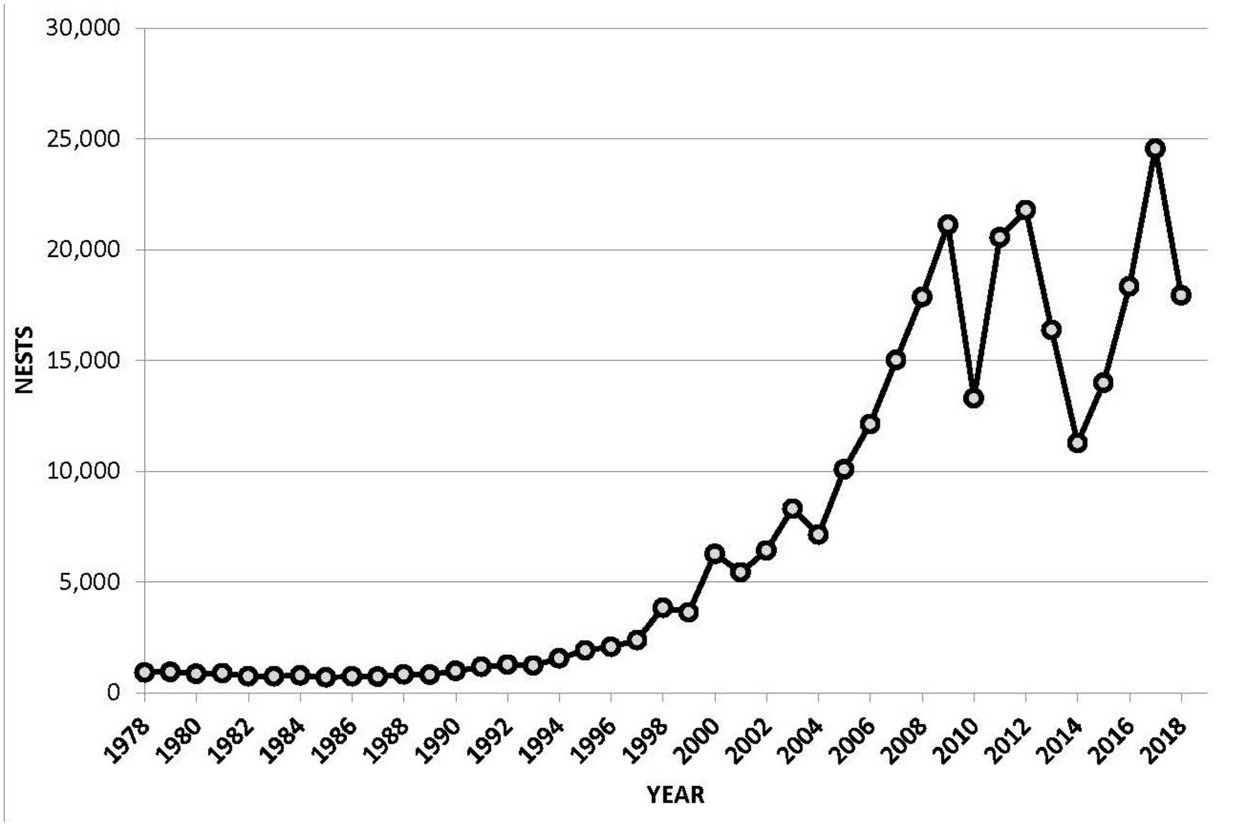

Figure 26. Kemp's ridley nest totals from Mexican beaches (Gladys Porter Zoo nesting database 2019)

Through modelling, Heppell et al. (2005) predicted the population is expected to increase at least 12-16% per year and could reach at least 10,000 females nesting on Mexico beaches by 2015. NMFS et al. (2011) produced an updated model that predicted the population to increase 19% per year and to attain at least 10,000 females nesting on Mexico beaches by 2011. Approximately 25,000 nests would be needed for an estimate of 10,000 nesters on the beach, based on an average 2.5 nests/nesting female. While counts did not reach 25,000 nests by 2015, it is clear that the population has increased over the long term. The increases in Kemp's ridley sea turtle nesting over the last 2 decades is likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of turtle excluder devices (TEDs), reduced trawling effort in Mexico and the United States, and possibly other changes in vital rates (Turtle Expert Working Group 1998; Turtle Expert Working Group 2000). While these results are encouraging, the species' limited range as well as low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental randomness, all factors which are often difficult to predict with any certainty. Additionally, the significant nesting declines observed in 2010 and 2013-2014 potentially indicate a serious population-level impact, and there is cause for concern regarding the ongoing recovery trajectory.

4.1.1.3.4 Threats

Kemp's ridley sea turtles face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in 4.1.1.1; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact Kemp's ridley sea turtles.

As Kemp's ridley sea turtles continue to recover and nesting arribadas[40] are increasingly established, bacterial and fungal pathogens in nests are also likely to increase. Bacterial and fungal pathogen impacts have been well documented in the large arribadas of the olive ridley at Nancite in Costa Rica (Mo 1988). In some years, and on some sections of the beach, the hatching success can be as low as 5% (Mo 1988). As the Kemp's ridley nest density at Rancho Nuevo and adjacent beaches continues to increase, appropriate monitoring of emergence success will be necessary to determine if there are any density-dependent effects.

Over the past 6 years, NMFS has documented (via the Sea Turtle Stranding and Salvage Network data, http://www.sefsc.noaa.gov/species/turtles/strandings.htm) elevated sea turtle strandings in the Northern Gulf of Mexico, particularly throughout the Mississippi Sound area. In the first 3 weeks of June 2010, over 120 sea turtle strandings were reported from Mississippi and Alabama waters, none of which exhibited any signs of external oiling to indicate effects associated with the DWH oil spill event. A total of 644 sea turtle strandings were reported in 2010 from Louisiana, Mississippi, and Alabama waters, 561 (87%) of which were Kemp's ridley sea turtles. During March through May of 2011, 267 sea turtle strandings were reported from Mississippi and Alabama waters alone. A total of 525 sea turtle strandings were reported in 2011 from Louisiana, Mississippi, and Alabama waters, with the majority (455) having occurred from March through July, 390 (86%) of which were Kemp's ridley sea turtles. During 2012, a total of 384 sea turtles were reported from Louisiana, Mississippi, and Alabama waters. Of these reported strandings, 343 (89%) were Kemp's ridley sea turtles. During 2014, a total of 285 sea turtles were reported from Louisiana, Mississippi, and Alabama waters, though the data is incomplete. Of these reported strandings, 229 (80%) were Kemp's ridley sea turtles. These stranding numbers are significantly greater than reported in past years; Louisiana, Mississippi, and Alabama waters reported 42 and 73 sea turtle strandings for 2008 and 2009, respectively. It should be noted that stranding coverage has increased considerably due to the DWH oil spill event.

Nonetheless, considering that strandings typically represent only a small fraction of actual mortality, these stranding events potentially represent a serious impact to the recovery and survival of the local sea turtle populations. While a definitive cause for these strandings has not been identified, necropsy results indicate a significant number of stranded turtles from these events likely perished due to forced submergence, which is commonly associated with fishery interactions (B. Stacy, NMFS, pers. comm. to M. Barnette, NMFS Southeast Region, March 2012). Yet, available information indicates fishery effort was extremely limited during the stranding events. The fact that 80% or more of all Louisiana, Mississippi, and Alabama stranded sea turtles in the past 5 years were Kemp's ridleys is notable; however, this could simply be a

[40] Arribada is the Spanish word for "arrival" and is the term used for massive synchronized nesting within the genus *Lepidochelys.*

function of the species' preference for shallow, inshore waters coupled with increased population abundance, as reflected in recent Kemp's ridley nesting increases.

In response to these strandings, and due to speculation that fishery interactions may be the cause, fishery observer effort was shifted to evaluate the inshore skimmer trawl fishery during the summer of 2012. During May-July of that year, observers reported 24 sea turtle interactions in the skimmer trawl fishery. All but a single sea turtle was identified as Kemp's ridleys (1 sea turtle was an unidentified hardshell turtle). Encountered sea turtles were all very small juvenile specimens, ranging from 7.6-19.0 inches (19.4-48.3 cm) curved carapace length. All sea turtles were released alive. The small average size of encountered Kemp's ridleys introduces a potential conservation issue, as over 50% of these reported sea turtles could potentially pass through the maximum 4-inch bar spacing of TEDs currently required in the shrimp fishery. Due to this issue, a proposed 2012 rule to require TEDs in the skimmer trawl fishery (77 FR 27411, Publication Date May 10, 2012) was not implemented. Based on anecdotal information, these interactions were a relatively new issue for the inshore skimmer trawl fishery. Given the nesting trends and habitat utilization of Kemp's ridley sea turtles, it is likely that fishery interactions in the Northern Gulf of Mexico may continue to be an issue of concern for the species, and one that may potentially slow the rate of recovery for Kemp's ridley sea turtles.

While oil spill impacts are discussed generally for all species in Section 4.1.1.1 of this Opinion, specific impacts of the DWH oil spill event on Kemp's ridley sea turtles are considered here. Kemp's ridleys experienced the greatest negative impact stemming from the DWH oil spill event of any sea turtle species. Impacts to Kemp's ridley sea turtles occurred to offshore small juveniles, as well as large juveniles and adults. Loss of hatchling production resulting from injury to adult turtles was also estimated for this species. Injuries to adult turtles of other species, such as loggerheads, certainly would have resulted in unrealized nests and hatchlings to those species as well. Yet, the calculation of unrealized nests and hatchlings was limited to Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of Mexico throughout their lives (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016).

A total of 217,000 small juvenile Kemp's ridleys (51.5% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. That means approximately half of all small juvenile Kemp's ridleys from the total population estimate of 430,000 oceanic small juveniles were exposed to oil. Furthermore, a large number of small juveniles were removed from the population, as up to 90,300 small juveniles Kemp's ridleys are estimated to have died as a direct result of the exposure. Therefore, as much as 20% of the small oceanic juveniles of this species were killed during that year. Impacts to large juveniles (>3 years old) and adults were also high. An estimated 21,990 such individuals were exposed to oil (about 22% of the total estimated population for those age classes); of those, 3,110 mortalities were estimated (or 3% of the population for those age classes). The loss of near-reproductive and reproductive-stage females would have contributed to some extent to the decline in total nesting abundance observed between 2011 and 2014. The estimated number of unrealized Kemp's ridley nests is between 1,300 and 2,000, which translates to between approximately

65,000 and 95,000 unrealized hatchlings (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). This is a minimum estimate, however, because the sublethal effects of the DWH oil spill event on turtles, their prey, and their habitats might have delayed or reduced reproduction in subsequent years, which may have contributed substantially to additional nesting deficits observed following the DWH oil spill event. These sublethal effects could have slowed growth and maturation rates, increased remigration intervals, and decreased clutch frequency (number of nests per female per nesting season). The nature of the DWH oil spill event effect on reduced Kemp's ridley nesting abundance and associated hatchling production after 2010 requires further evaluation. It is clear that the DWH oil spill event resulted in large losses to the Kemp's ridley population across various age classes, and likely had an important population-level effect on the species. Still, we do not have a clear understanding of those impacts on the population trajectory for the species into the future.

4.1.1.4 Leatherback Sea Turtle

The leatherback sea turtle was listed as endangered throughout its entire range , (35 FR 8491, Publication Date June 2, 1970) under the Endangered Species Conservation Act of 1969.

4.1.1.4.1 Species Description and Distribution

The leatherback is the largest sea turtle in the world, with a curved carapace length that often exceeds 5 ft (150 cm) and front flippers that can span almost 9 ft (270 cm) (NMFS and USFWS 1998). Mature males and females can reach lengths of over 6 ft (2 m) and weigh close to 2,000 lb (900 kg). The leatherback does not have a bony shell. Instead, its shell is approximately 1.5 inch (4 cm) thick and consists of a leathery, oil-saturated connective tissue overlaying loosely interlocking dermal bones. The ridged shell and large flippers help the leatherback during its long-distance trips in search of food.

Unlike other sea turtles, leatherbacks have several unique traits that enable them to live in cold water. For example, leatherbacks have a countercurrent circulatory system (Greer Jr. et al. 1973),[41] a thick layer of insulating fat (Davenport et al. 1990a; Goff and Lien 1988), gigantothermy (Paladino et al. 1990),[42] and they can increase their body temperature through increased metabolic activity (Bostrom and Jones 2007; Southwood et al. 2005). These adaptations allow leatherbacks to be comfortable in a wide range of temperatures, which helps them to travel further than any other sea turtle species (NMFS and USFWS 1995). For example, a leatherback may swim more than 6,000 miles (10,000 km) in a single year (Benson et al. 2007a; Benson et al. 2011; Eckert 2006; Eckert et al. 2006). They search for food between latitudes 71°N and 47°S in all oceans, and travel extensively to and from their tropical nesting beaches. In the Atlantic Ocean, leatherbacks have been recorded as far north as Newfoundland, Canada, and Norway, and as far south as Uruguay, Argentina, and South Africa (NMFS 2001b).

[41] Countercurrent circulation is a highly efficient means of minimizing heat loss through the skin's surface because heat is recycled. For example, a countercurrent circulation system often has an artery containing warm blood from the heart surrounded by a bundle of veins containing cool blood from the body's surface. As the warm blood flows away from the heart, it passes much of its heat to the colder blood returning to the heart via the veins. This conserves heat by recirculating it back to the body's core.
[42] "Gigantothermy" refers to a condition when an animal has relatively high volume compared to its surface area, and as a result, it loses less heat.

While leatherbacks will look for food in coastal waters, they appear to prefer the open ocean at all life stages (Heppell et al. 2003b). Leatherbacks have pointed tooth-like cusps and sharp-edged jaws that are adapted for a diet of soft-bodied prey such as jellyfish and salps. A leatherback's mouth and throat also have backward-pointing spines that help retain jelly-like prey. Leatherbacks' favorite prey are jellies (e.g., medusae, siphonophores, and salps), which commonly occur in temperate and northern or sub-arctic latitudes and likely has a strong influence on leatherback distribution in these areas (Plotkin 2003). Leatherbacks are known to be deep divers, with recorded depths in excess of a half-mile (Eckert et al. 1989a), but they may also come into shallow waters to locate prey items.

Genetic analyses using microsatellite markers along with mitochondrial DNA and tagging data indicate there are 7 groups or breeding populations in the Atlantic Ocean: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (Turtle Expert Working Group 2007). General differences in migration patterns and foraging grounds may occur between the 7 nesting assemblages, although data to support this is limited in most cases.

4.1.1.4.2 Life History Information

The leatherback life cycle is broken into several stages: (1) egg/hatchling, (2) post-hatchling, (3) juvenile, (4) subadult, and (5) adult. Leatherbacks are a long-lived species that delay age of maturity, have low and variable survival in the egg and juvenile stages, and have relatively high and constant annual survival in the subadult and adult life stages (Chaloupka 2002; Crouse 1999; Heppell et al. 1999; Heppell et al. 2003b; Spotila et al. 1996; Spotila et al. 2000). While a robust estimate of the leatherback sea turtle's life span does not exist, the current best estimate for the maximum age is 43 (Avens et al. 2009). It is still unclear when leatherbacks first become sexually mature. Using skeletochronological data, Avens et al. (2009) estimated that leatherbacks in the western North Atlantic may not reach maturity until 29 years of age, which is longer than earlier estimates of 2-3 years by Pritchard and Trebbau (1984), of 3-6 years by Rhodin (1985), of 13-14 years for females by Zug and Parham (1996), and 12-14 years for leatherbacks nesting in the U.S. Virgin Islands by Dutton et al. (2005). A more recent study that examined leatherback growth rates estimated an age at maturity of 16.1 years (Jones et al. 2011).

The average size of reproductively active females in the Atlantic is generally 5-5.5 ft (150-162 cm) curved carpace length (Benson et al. 2007b; Hirth et al. 1993; Starbird and Suarez 1994). Still, females as small as 3.5-4 ft (105-125 cm) CCL have been observed nesting at various sites (Stewart et al. 2007).

Female leatherbacks typically nest on sandy, tropical beaches at intervals of 2-4 years (García-Muñoz and Sarti 2000; McDonald and Dutton 1996; Spotila et al. 2000). Unlike other sea turtle species, female leatherbacks do not always nest at the same beach year after year; some females may even nest at different beaches during the same year (Dutton et al. 2005; Eckert et al. 1989b; Keinath and Musick 1993; Steyermark et al. 1996). Individual female leatherbacks have been observed with fertility spans as long as 25 years (Hughes 1996). Females usually lay up to 10 nests during the 3-6 month nesting season (March through July in the United States), typically 8-12 days apart, with 100 eggs or more per nest (Eckert et al. 2012; Eckert et al. 1989b; Maharaj

2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). Yet, up to approximately 30% of the eggs may be infertile (Eckert et al. 1989b; Eckert et al. 1984; Maharaj 2004; Matos 1986; Stewart and Johnson 2006; Tucker 1988). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately 50% worldwide (Eckert et al. 2012), which is lower than the greater than 80% reported for other sea turtle species (Miller 1997). In the United States, the emergent success is higher at 54-72% (Eckert and Eckert 1990; Stewart and Johnson 2006; Tucker 1988). Thus the number of hatchlings in a given year may be less than the total number of eggs produced in a season. Eggs hatch after 60-65 days, and the hatchlings have white striping along the ridges of their backs and on the edges of the flippers. Leatherback hatchlings weigh approximately 1.5-2 ounces (40-50 grams), and have lengths of approximately 2-3 inches (51-76 mm), with fore flippers as long as their bodies. Hatchlings grow rapidly, with reported growth rates for leatherbacks from 2.5-27.6 inches (6-70 cm) in length, estimated at 12.6 inches (32 cm) per year (Jones et al. 2011).

In the Atlantic, the sex ratio appears to be skewed toward females. The Turtle Expert Working Group reports that nearshore and onshore strandings data from the U.S. Atlantic and Gulf of Mexico coasts indicate that 60% of strandings were females (Turtle Expert Working Group 2007). Those data also show that the proportion of females among adults (57%) and juveniles (61%) was also skewed toward females in these areas (Turtle Expert Working Group 2007). James et al. (2007) collected size and sex data from large subadult and adult leatherbacks off Nova Scotia and also concluded a bias toward females at a rate of 1.86:1.

The survival and mortality rates for leatherbacks are difficult to estimate and vary by location. For example, the annual mortality rate for leatherbacks that nested at Playa Grande, Costa Rica, was estimated to be 34.6% in 1993-1994, and 34.0% in 1994-1995 (Spotila et al. 2000). In contrast, leatherbacks nesting in French Guiana and St. Croix had estimated annual survival rates of 91% (Rivalan et al. 2005) and 89% (Dutton et al. 2005), respectively. For the St. Croix population, the average annual juvenile survival rate was estimated to be approximately 63% and the total survival rate from hatchling to first year of reproduction for a female was estimated to be between 0.4% and 2%, assuming age at first reproduction is between 9-13 years (Eguchi et al. 2006). Spotila et al. (1996) estimated first-year survival rates for leatherbacks at 6.25%.

Migratory routes of leatherbacks are not entirely known; however, recent information from satellite tags have documented long travels between nesting beaches and foraging areas in the Atlantic and Pacific Ocean basins (Benson et al. 2007a; Benson et al. 2011; Eckert 2006; Eckert et al. 2006; Ferraroli et al. 2004; Hays et al. 2004; James et al. 2005). Leatherbacks nesting in Central America and Mexico travel thousands of miles through tropical and temperate waters of the South Pacific (Eckert and Sarti 1997; Shillinger et al. 2008). Data from satellite tagged leatherbacks suggest that they may be traveling in search of seasonal aggregations of jellyfish (Benson et al. 2007b; Bowlby et al. 1994; Graham 2009; Shenker 1984; Starbird et al. 1993; Suchman and Brodeur 2005).

4.1.1.4.3 Status and Population Dynamics

The status of the Atlantic leatherback population has been less clear than the Pacific population, which has shown dramatic declines at many nesting sites (Martínez et al. 2007; Santidrián Tomillo et al. 2007; Spotila et al. 2000). This uncertainty resulted from inconsistent beach and aerial surveys, cycles of erosion, and reformation of nesting beaches in the Guianas (representing the largest nesting area). Leatherbacks also show a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species. Coordinated efforts of data collection and analyses by the leatherback Turtle Expert Working Group helped to clarify the understanding of the Atlantic population status up through the early 2000's (Turtle Expert Working Group 2007). However, additional information for the Northwest Atlantic population has more recently shown declines in that population as well, contrary to what earlier information indicated (Northwest Atlantic Leatherback Working Group 2018). A full status review covering leatherback status and trends for all populations worldwide is being finalized.

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (Turtle Expert Working Group 2007). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with most of the nesting occurring in the Guianas and Trinidad. The Southern Caribbean/Guianas stock of leatherbacks was designated after genetics studies indicated that animals from the Guianas (and possibly Trinidad) should be viewed as a single population. Using nesting females as a proxy for population, the Turtle Expert Working Group (2007) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate. The Turtle Expert Working Group observed positive growth within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (Turtle Expert Working Group 2007). More specifically, Tiwari et al. (2013) report an estimated 3-generation abundance change of +3%, +20,800%, +1,778%, and +6% in Trinidad, Guyana, Suriname, and French Guiana, respectively. However, subsequent analysis using data up through 2017 has shown decreases in this stock, with an annual geometric mean decline of 10.43% over what they described as the short term (2008-2017) and a long-term (1990-2017) annual geometric mean decline of 5% (Northwest Atlantic Leatherback Working Group 2018).

Researchers believe the cyclical pattern of beach erosion and then reformation has affected leatherback nesting patterns in the Guianas. For example, between 1979 and 1986, the number of leatherback nests in French Guiana had increased by about 15% annually (NMFS 2001b). This increase was then followed by a nesting decline of about 15% annually. This decline corresponded with the erosion of beaches in French Guiana and increased nesting in Suriname. This pattern suggests that the declines observed since 1987 might actually be a part of a nesting cycle that coincides with cyclic beach erosion in Guiana (Schulz 1975). Researchers think that the cycle of erosion and reformation of beaches may have changed where leatherbacks nest throughout this region. The idea of shifting nesting beach locations was supported by increased nesting in Suriname,[43] while the number of nests was declining at beaches in Guiana (Hilterman et al. 2003). Though this information suggested the long-term trend for the overall Suriname and French Guiana population was increasing. A more recent cycle of nesting declines from 2008-2017, as high at 31% annual decline in the Awala-Yalimapo area of French Guiana and almost 20% annual declines in Guyana, has changed the long-term nesting trends in the region to negative as described above (Northwest Atlantic Leatherback Working Group 2018).

[43] Leatherback nesting in Suriname increased by more than 10,000 nests per year since 1999 with a peak of 30,000 nests in 2001.

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. Across the Western Caribbean, nesting is most prevalent in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coastline of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from index nesting beaches in Tortuguero, Gandoca, and Pacuaré in Costa Rica indicate that the nesting population likely was not growing over the 1995-2005 time series (Turtle Expert Working Group 2007). Other modeling of the nesting data for Tortuguero indicates a possible 67.8% decline between 1995 and 2006 (Troëng et al. 2007). Tiwari et al. (2013) report an estimated 3-generation abundance change of -72%, -24%, and +6% for Tortuguero, Gandoca, and Pacuare, respectively. Further decline of almost 6% annual geometric mean from 2008-2017 reflects declines in nesting beaches throughout this stock (Northwest Atlantic Leatherback Working Group 2018).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, St. Croix (U.S. Virgin Islands), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (Turtle Expert Working Group 2007). Tiwari et al. (2013) report an estimated three-generation abundance change of -4% and +5,583% at Culebra and Fajardo, respectively. At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has varied from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (Turtle Expert Working Group 2007). From 2006-2010, Tiwari et al. (2013) report an annual growth rate of +7.5% in St. Croix and a three-generation abundance change of +1,058%. Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2% between 1994 and 2004 (Turtle Expert Working Group 2007). The nesting trend reversed course later, with an annual geometric mean decline of 10% from 2008-2017 driving the long-term trend (1990-2017) down to a 2% annual decline (Northwest Atlantic Leatherback Working Group 2018).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the Turtle Expert Working Group (2007) estimated a significant annual nesting growth rate of 1.17% between 1989 and 2005. Florida Fish and Wildlife Conservation Commission Index Nesting Beach Survey Data generally indicates biennial peaks in nesting abundance beginning in 2007 (Figure 27 and Table 28). A similar pattern was also observed statewide (Table 28). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting. Overall, the trend showed growth on Florida's east coast beaches. Tiwari et al. (2013) report an annual growth rate of 9.7% and a 3-generation abundance change of +1,863%. However, in recent years nesting has declined on Florida beaches, with 2017 hitting a decade-low number, with a partial rebound in 2018. Similar patterns are being seen in other nesting beaches of the northwest Atlantic. A status review is currently (2018) underway to analyze leatherback status and trends worldwide.

Table 28. Number of Leatherback Sea Turtle Nests in Florida

Nests Recorded	2011	2012	2013	2014	2015	2016	2017	2018
Index Nesting Beaches	625	515	322	641	489	319	205	316
Statewide	1,653	1,712	896	1,604	1,493	1,054	663	949

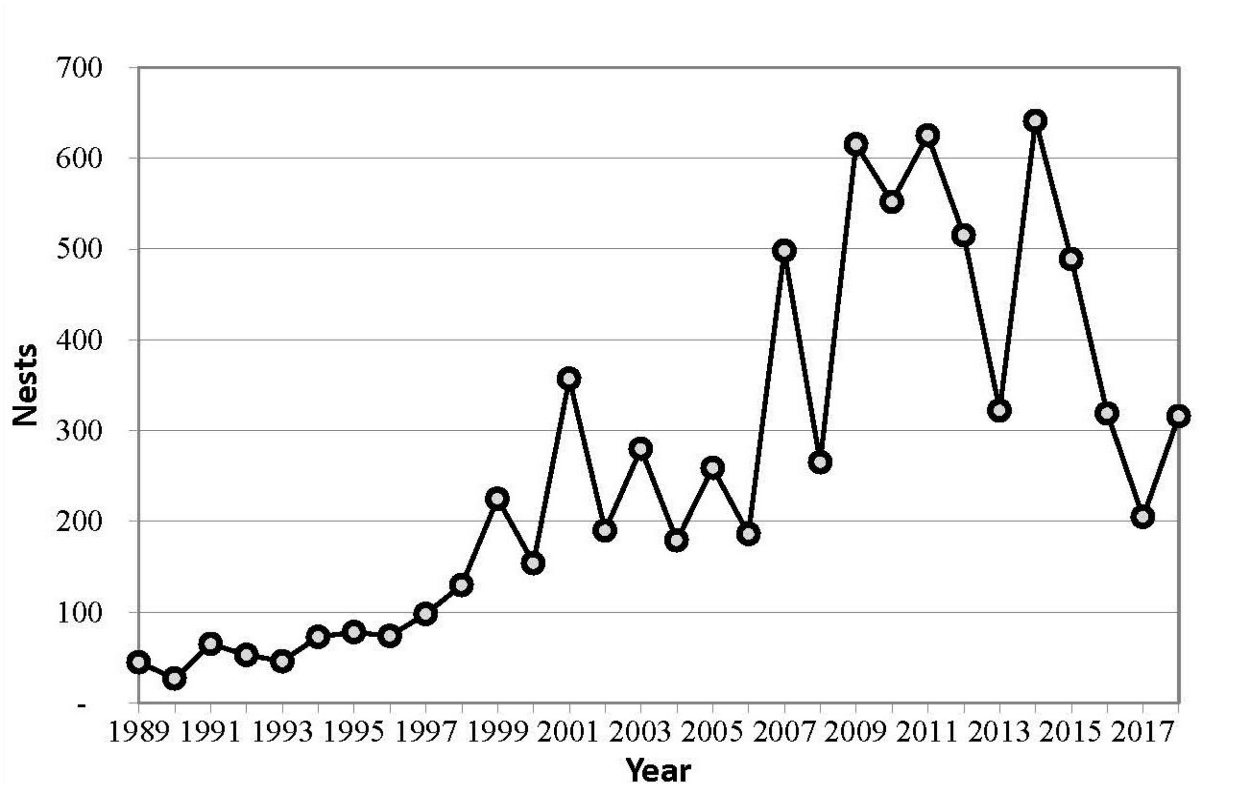

Figure 27. Leatherback sea turtle nesting at Florida index beaches since 1989

The West African nesting stock of leatherbacks is large and important, but it is a mostly unstudied aggregation. Nesting occurs in various countries along Africa's Atlantic coast, but much of the nesting is undocumented and the data are inconsistent. Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in a single season (Fretey et al. 2007). Fretey et al. (2007) provide detailed information about other known nesting beaches and survey efforts along the Atlantic African coast. Because of the lack of consistent effort and minimal available data, trend analyses were not possible for this stock (Turtle Expert Working Group 2007).

Two other small but growing stocks nest on the beaches of Brazil and South Africa. Based on the data available, Turtle Expert Working Group (2007) determined that between 1988 and 2003, there was a positive annual average growth rate between 1.07% and 1.08% for the Brazilian stock. Turtle Expert Working Group (2007) estimated an annual average growth rate between 1.04% and 1.06% for the South African stock.

Because the available nesting information is inconsistent, it is difficult to estimate the total population size for Atlantic leatherbacks. Spotila et al. (1996) characterized the entire Western Atlantic population as stable at best and estimated a population of 18,800 nesting females.

Spotila et al. (1996) further estimated that the adult female leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, was about 27,600 (considering both nesting and interesting females), with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the Turtle Expert Working Group (2007). The Turtle Expert Working Group (2007) also determined that at of the time of their publication, leatherback sea turtle populations in the Atlantic were all stable or increasing with the exception of the Western Caribbean and West Africa populations. A later review by NMFS and USFWS (2013) suggested the leatherback nesting population was stable in most nesting regions of the Atlantic Ocean. However, as described earlier, the northwest Atlantic population has experienced declines over the near term (2008-2017), often severe enough to reverse the longer term trends to negative where increases had previously been seen (Northwest Atlantic Leatherback Working Group 2018). Given the relatively large size of the northwest Atlantic population, it is likely that the overall Atlantic leatherback trend is no longer increasing NMFS and USFWS (2013).

4.1.1.4.4 Threats

Leatherbacks face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.1.1.1 of this Opinion; the remainder of this section will expand on a few of the aforementioned threats and how they may specifically impact leatherback sea turtles.

Of all sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, especially gillnet and pot/trap lines. This vulnerability may be because of their body type (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, their method of locomotion, and/or their attraction to the lightsticks used to attract target species in longline fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine and many other stranded individuals exhibited evidence of prior entanglement (Dwyer et al. 2003). Zug and Parham (1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment from intense egg harvesting in some areas has caused a sharp decline in leatherback sea turtle populations. This represents a significant threat to survival and recovery of the species worldwide.

Leatherback sea turtles may also be more susceptible to marine debris ingestion than other sea turtle species due to their predominantly pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding and migratory purposes (Lutcavage et al. 1997; Shoop and Kenney 1992). The stomach contents of leatherback sea turtles revealed that a substantial percentage (33.8% or 138 of 408 cases examined) contained some form of plastic debris (Mrosovsky et al. 2009). Blocking of the gut by plastic to an extent that could have caused death was evident in 8.7% of all leatherbacks that ingested plastic (Mrosovsky et al. 2009). Mrosovsky et al. (2009) also note that in a number of cases, the ingestion of plastic may not cause death outright, but could cause the animal to absorb fewer nutrients from food, eat less in general, etc.– factors which could cause other adverse effects. The presence of plastic in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and forms of debris such a plastic bags (Mrosovsky et al. 2009). Balazs (1985) speculated that the plastic object might resemble a food item by its shape, color, size, or even movement as it drifts about, and therefore induce a feeding response in leatherbacks.

As discussed in Section 4.1.1.1 of this Opinion, global climate change can be expected to have various impacts on all sea turtles, including leatherbacks. Global climate change is likely to also influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007c). Several studies have shown leatherback distribution is influenced by jellyfish abundance (Houghton et al. 2006; Witt et al. 2007; Witt et al. 2006); however, more studies need to be done to monitor how changes to prey items affect distribution and foraging success of leatherbacks so population-level effects can be determined.

While oil spill impacts are discussed generally for all species in Section 4.1.1.1 of this Opinion, specific impacts of the DWH oil spill on leatherback sea turtles are considered here. Available information indicates leatherback sea turtles (along with hawksbill turtles) were likely directly affected by the oil spill. Leatherbacks were documented in the spill area, but the number of affected leatherbacks was not estimated due to a lack of information compared to other species. But given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (Turtle Expert Working Group 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, it was concluded that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died. Potential DWH-related impacts to leatherback sea turtles include direct oiling or contact with dispersants from surface and subsurface oil and dispersants, inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred. Although adverse impacts likely occurred to leatherbacks, the relative proportion of the population that is expected to have been exposed to and directly impacted by the DWH event may be relatively low. Thus, a population-level impact may not have occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

4.1.1.5 Loggerhead Sea Turtle – Northwest Atlantic DPS

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. NMFS and USFWS subsequently published a Final Rule which designated 9 DPSs for loggerhead sea turtles (76 FR 58868, Publication Date September 22, 2011). This rule listed the following DPSs: (1) Northwest Atlantic Ocean (threatened), (2) Northeast Atlantic Ocean (endangered), (3) South Atlantic Ocean (threatened), (4) Mediterranean Sea (endangered), (5) North Pacific Ocean (endangered), (6) South Pacific Ocean (endangered), (7) North Indian Ocean (endangered), (8) Southeast Indo-Pacific Ocean (endangered), and (9) Southwest Indian Ocean (threatened). The NWA DPS is the only one that occurs within the action area, and therefore it is the only one considered in this Opinion.

4.1.1.5.1 Species Description and Distribution

Loggerheads are large sea turtles. Adults in the southeast United States average about 3 ft (92 cm) long, measured as a straight carapace length, and weigh approximately 255 lb (116 kg) (Ehrhart and Yoder 1978). Adult and subadult loggerhead sea turtles typically have a light yellow plastron and a reddish brown carapace covered by non-overlapping scutes that meet along seam lines. They typically have 11 or 12 pairs of marginal scutes, 5 pairs of costals, 5 vertebrals, and a nuchal (precentral) scute that is in contact with the first pair of costal scutes (Dodd Jr. 1988).

The loggerhead sea turtle inhabits continental shelf and estuarine environments throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd Jr. 1988). Habitat uses within these areas vary by life stage. Juveniles are omnivorous and forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd Jr. 1988). Subadult and adult loggerheads are primarily found in coastal waters and eat benthic invertebrates such as mollusks and decapod crustaceans in hardbottom habitats.

The majority of loggerhead nesting occurs at the western rims of the Atlantic and Indian Oceans concentrated in the north and south temperate zones and subtropics (National Research Council 1990a). For the NWA DPS, most nesting occurs along the coast of the United States, from southern Virginia to Alabama. Additional nesting beaches for this DPS are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba (Moncada-Gavilán 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

Non-nesting, adult female loggerheads are reported throughout the U.S. Atlantic, Gulf of Mexico, and Caribbean Sea. Little is known about the distribution of adult males who are seasonally abundant near nesting beaches. Aerial surveys suggest that loggerheads as a whole are distributed in U.S. waters as follows: 54% off the southeast U.S. coast, 29% off the northeast U.S. coast, 12% in the eastern Gulf of Mexico, and 5% in the western Gulf of Mexico (Turtle Expert Working Group 1998).

Within the NWA DPS, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf Coast of Florida. Previous Section 7 analyses have recognized at least 5 western Atlantic subpopulations, divided geographically as follows: (1) a Northern nesting subpopulation, occurring from North Carolina to northeast Florida at about 29ºN; (2) a South Florida nesting subpopulation, occurring from 29°N on the east coast of the state to Sarasota on the west coast; (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez M. 1990; Turtle Expert Working Group 2000); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS 2001b).

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles concluded that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida

Peninsula. It also concluded that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the recovery plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are as follows: (1) the Northern Recovery Unit (Florida/Georgia border north through southern Virginia), (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The recovery plan concluded that all recovery units are essential to the recovery of the species. Although the recovery plan was written prior to the listing of the NWA DPS, the recovery units for what was then termed the Northwest Atlantic population apply to the NWA DPS.

4.1.1.5.2 Life History Information

The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, which include the ecosystems those stages generally use: (1) egg (terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone[44]), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting female (terrestrial zone) (NMFS and USFWS 2008). Loggerheads are long-lived animals. They reach sexual maturity between 20-38 years of age, although age of maturity varies widely among populations (Frazer and Ehrhart 1985; NMFS 2001b). The annual mating season occurs from late March to early June, and female turtles lay eggs throughout the summer months. Females deposit an average of 4.1 nests within a nesting season (Murphy and Hopkins 1984), but an individual female only nests every 3.7 years on average (Tucker 2010). Each nest contains an average of 100-126 eggs (Dodd Jr. 1988) which incubate for 42-75 days before hatching (NMFS and USFWS 2008). Loggerhead hatchlings are 1.5-2 inches long and weigh about 0.7 ounces (20 grams).

As post-hatchlings, loggerheads hatched on U.S. beaches enter the “oceanic juvenile” life stage, migrating offshore and becoming associated with *Sargassum* habitats, driftlines, and other convergence zones (Carr 1986; Conant et al. 2009; Witherington 2002). Oceanic juveniles grow at rates of 1-2 inches (2.9-5.4 cm) per year (Bjorndal et al. 2003; Snover 2002) over a period as long as 7-12 years (Bolten et al. 1998) before moving to more coastal habitats. Studies have suggested that not all loggerhead sea turtles follow the model of circumnavigating the North Atlantic Gyre as pelagic juveniles, followed by permanent settlement into benthic environments (Bolten and Witherington 2003; Laurent et al. 1998). These studies suggest some turtles may either remain in the oceanic habitat in the North Atlantic longer than hypothesized, or they move back and forth between oceanic and coastal habitats interchangeably (Witzell 2002). Stranding records indicate that when immature loggerheads reach 15-24 in (40-60 cm) straight carapace length, they begin to reside in coastal inshore waters of the continental shelf throughout the U.S. Atlantic and Gulf of Mexico (Witzell 2002).

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[44] Neritic refers to the nearshore marine environment from the surface to the sea floor where water depths do not exceed 200 m.

After departing the oceanic zone, neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Estuarine waters of the United States, including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, Florida Bay, as well as numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads (Conant et al. 2009).

Like juveniles, non-nesting adult loggerheads also use the neritic zone. However, these adult loggerheads do not use the relatively enclosed shallow-water estuarine habitats with limited ocean access as frequently as juveniles. Areas such as Pamlico Sound, North Carolina, and the Indian River Lagoon, Florida, are regularly used by juveniles but not by adult loggerheads. Adult loggerheads do tend to use estuarine areas with more open ocean access, such as the Chesapeake Bay in the U.S. mid-Atlantic. Shallow-water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads (Conant et al. 2009).

Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has also been documented (Hawkes et al. 2007) Georgia Department of Natural Resources, unpublished data; South Carolina Department of Natural Resources, unpublished data). Satellite telemetry has identified the shelf waters along the west Florida coast, The Bahamas, Cuba, and the Yucatán Peninsula as important resident areas for adult female loggerheads that nest in Florida (Foley et al. 2008; Girard et al. 2009; Hart et al. 2012). The southern edge of the Grand Bahama Bank is important habitat for loggerheads nesting on the Cay Sal Bank in The Bahamas, but nesting females are also resident in the bights of Eleuthera, Long Island, and Ragged Islands. They also reside in Florida Bay in the United States, and along the north coast of Cuba (A. Bolten and K. Bjorndal, University of Florida, unpublished data). Moncada et al. (2010) report the recapture of 5 adult female loggerheads in Cuban waters originally flipper-tagged in Quintana Roo, Mexico, which indicates that Cuban shelf waters likely also provide foraging habitat for adult females that nest in Mexico.

Status and Population Dynamics
A number of stock assessments and similar reviews (Conant et al. 2009; Heppell et al. 2003a; NMFS 2001b; NMFS 2009a; NMFS and USFWS 2008; Turtle Expert Working Group 1998; Turtle Expert Working Group 2000; Turtle Expert Working Group 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

Numbers of nests and nesting females can vary widely from year to year. Nesting beach surveys, though, can provide a reliable assessment of trends in the adult female population, due to the strong nest site fidelity of female loggerhead sea turtles, as long as such studies are sufficiently long and survey effort and methods are standardized (e.g., (NMFS and USFWS 2008). NMFS and USFWS (2008) concluded that the lack of change in 2 important demographic parameters of loggerheads, remigration interval and clutch frequency, indicate that time series on numbers of nests can provide reliable information on trends in the female population.

Peninsular Florida Recovery Unit

The Peninsular Florida Recovery Unit is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed an average of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (NMFS and USFWS 2008). The statewide estimated total for 2017 was 96,912 nests (Florida Fish and Wildlife Research Institute nesting database).

In addition to the total nest count estimates, the Florida Fish and Wildlife Research Institute uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. This provides a better tool for understanding the nesting trends (Figure 28). Florida Fish and Wildlife Research Institute performed a detailed analysis of the long-term loggerhead index nesting data (1989-2017; http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trend/). Over that time period, 3 distinct trends were identified. From 1989-1998, there was a 24% increase that was followed by a sharp decline over the subsequent 9 years. A large increase in loggerhead nesting has occurred since, as indicated by the 71% increase in nesting over the 10-year period from 2007 and 2016. Nesting in 2016 also represented a new record for loggerheads on the core index beaches. Florida Fish and Wildlife Research Institute examined the trend from the 1998 nesting high through 2016 and found that the decade-long post-1998 decline was replaced with a slight but nonsignificant increasing trend. Looking at the data from 1989 through 2016, Florida Fish and Wildlife Research Institute concluded that there was an overall positive change in the nest counts although it was not statistically significant due to the wide variability between 2012-2016 resulting in widening confidence intervals (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trend/). Nesting at the core index beaches declined in 2017 to 48,033, which is still the 4th highest total since 2001.

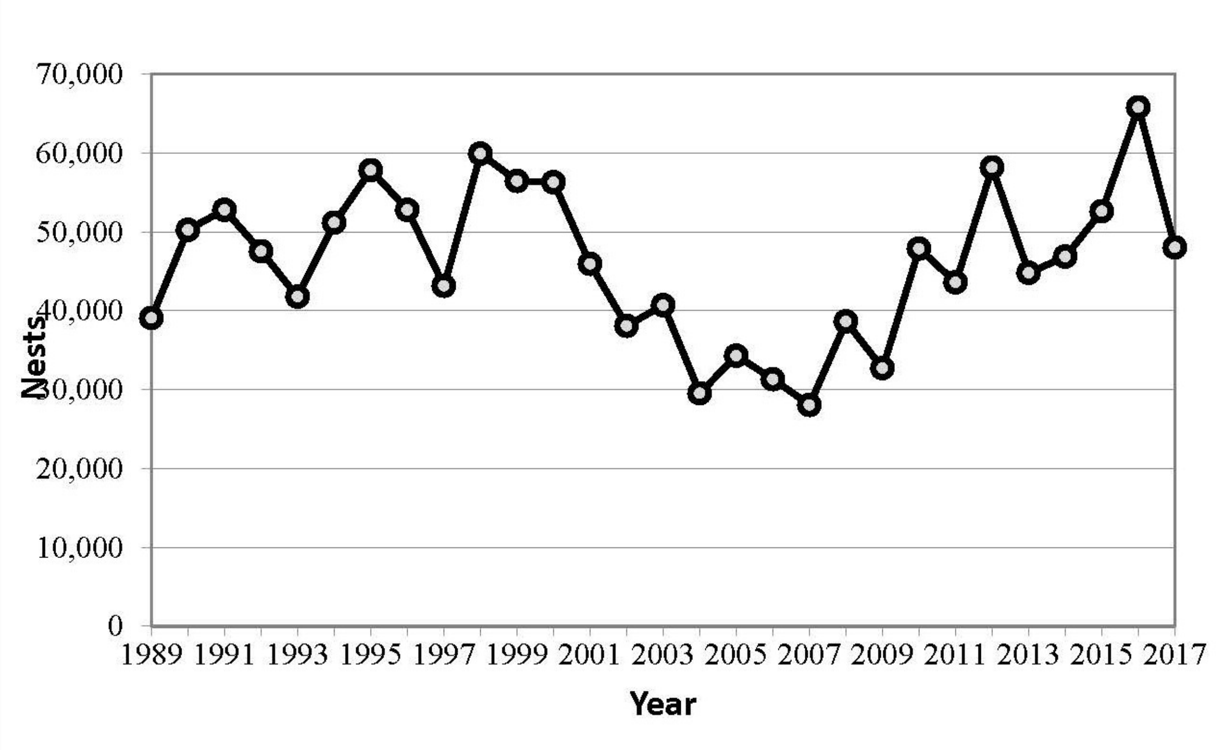

Figure 28. Loggerhead sea turtle nesting at Florida index beaches since 1989

Northern Recovery Unit
Annual nest totals from beaches within the Northern Recovery Unit averaged 5,215 nests from 1989-2008, a period of near-complete surveys of Northern Recovery Unit nesting beaches (Georgia Department of Natural Resources unpublished data, North Carolina Wildlife Resources Commission unpublished data, South Carolina Department of Natural Resources unpublished data), and represent approximately 1,272 nesting females per year, assuming 4.1 nests per female (Murphy and Hopkins 1984). The loggerhead nesting trend from daily beach surveys showed a significant decline of 1.3% annually from 1989-2008. Nest totals from aerial surveys conducted by South Carolina Department of Natural Resources showed a 1.9% annual decline in nesting in South Carolina from 1980-2008. Overall, there are strong statistical data to suggest the Northern Recovery Unit had experienced a long-term decline over that period of time.

Data since that analysis (Table 29) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, Georgia Department of Natural Resources press release, http://www.georgiawildlife.com/node/3139). South Carolina and North Carolina nesting have also begun to shift away from the past declining trend. Loggerhead nesting in Georgia, South Carolina, and North Carolina all broke records in 2015 and then topped those records again in 2016. Nesting in 2017 declined relative to 2016, back to levels seen in 2013 and 2015.

Table 29. Total Number of Northern Recovery Unit Loggerhead Nests (Georgia Department of Natural Resources, South Carolina Department of Natural Resources, and North Carolina Wildlife Resources Commission nesting datasets compiled at Seaturtle.org)

Nests Recorded	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Georgia	1,649	998	1,760	1,992	2,241	2,289	1,196	2,319	3,265	2,155
South Carolina	4,500	2,182	3,141	4,015	4,615	5,193	2,083	5,104	6,443	5,232
North Carolina	841	302	856	950	1,074	1,260	542	1,254	1,612	1,195
Total	6,990	3,472	5,757	6,957	7,930	8,742	3,821	8,677	11,320	8,582

South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2013, with a subsequent steep drop in 2014. Nesting then rebounded in 2015 and 2016, setting new highs each of those years. Nesting in 2017 dropped back down from the 2016 high, but was still the second highest on record (Figure 29).

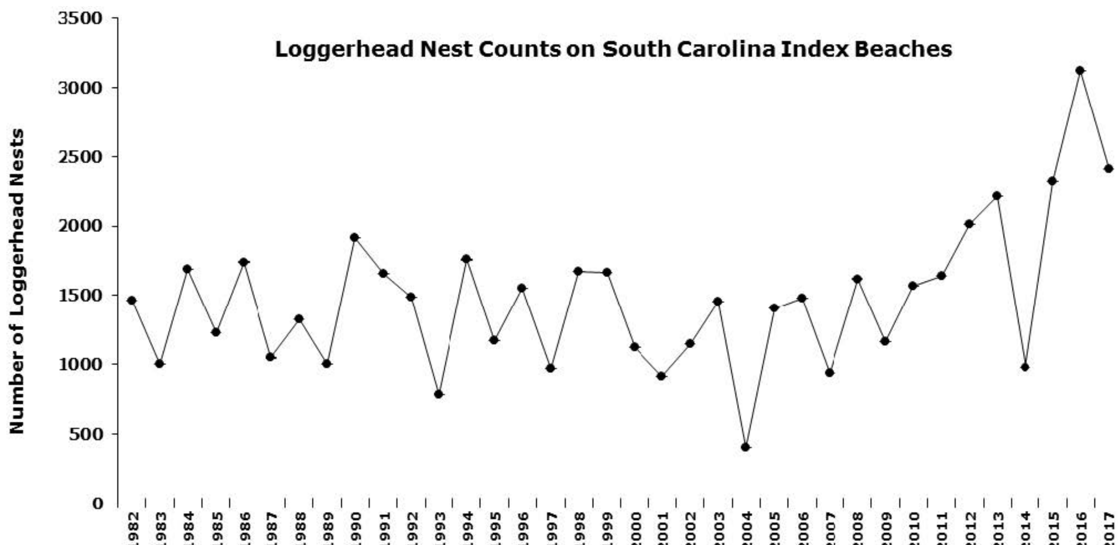

Figure 29. South Carolina index nesting beach counts for loggerhead sea turtles
Figure from the South Carolina Department of Natural Resources website: http://www.dnr.sc.gov/seaturtle/nest.htm)

Other Northwest Atlantic DPS Recovery Units
The remaining 3 recovery units—Dry Tortugas, Northern Gulf of Mexico, and Greater Caribbean—are much smaller nesting assemblages, but they are still considered essential to the continued existence of the species. Nesting surveys for the Dry Tortugus recovery unit are conducted as part of Florida's statewide survey program. Survey effort was relatively stable during the 9-year period from 1995-2004, although the 2002 year was missed. Nest counts ranged from 168-270, with a mean of 246, but there was no detectable trend during this period (NMFS and USFWS 2008). Nest counts for the Northern Gulf of Mexico recovery unit are focused on index beaches rather than all beaches where nesting occurs. Analysis of the 12-year dataset (1997-2008) of index nesting beaches in the area shows a statistically significant declining trend of 4.7% annually. Nesting on the Florida Panhandle index beaches, which represents the majority of Northern Gulf of Mexico recovery unit nesting, had shown a large increase in 2008, but then declined again in 2009 and 2010 before rising back to a level similar to the 2003-2007 average in 2011. Nesting survey effort has been inconsistent among the Greater Caribbean recovery unit nesting beaches, and no trend can be determined for this subpopulation (NMFS and USFWS 2008). Zurita et al. (2003) found a statistically significant increase in the number of nests on 7 of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. Nonetheless, nesting has declined since 2001, and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008).

In-water Trends
Nesting data are the best current indicator of sea turtle population trends, but in-water data also provide some insight. In-water research suggests the abundance of neritic juvenile loggerheads is steady or increasing. Although Ehrhart et al. (2007) found no significant regression-line trend in a long-term dataset, researchers have observed notable increases in CPUE (Arendt et al. 2009; Ehrhart et al. 2007; Epperly et al. 2007). Researchers believe that this increase in CPUE is likely linked to an increase in juvenile abundance, although it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. Bjorndal et al. (2005), cited in NMFS and USFWS (2008), caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern United States may be due to increased abundance of the largest oceanic/neritic juveniles (historically referred to as small benthic juveniles), which could indicate a relatively large number of individuals around the same age may mature in the near future (Turtle Expert Working Group 2009). In-water studies throughout the eastern United States, however, indicate a substantial decrease in the abundance of the smallest oceanic/neritic juvenile loggerheads, a pattern corroborated by stranding data (Turtle Expert Working Group 2009).

4.1.1.5.3 Population Estimate

The NMFS Southeast Fisheries Science Center developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS 2009a). The model uses the range of published information for the various parameters including mortality by stage, stage duration (years in a stage), and fecundity parameters such as eggs per nest, nests per nesting female, hatchling emergence success, sex ratio, and remigration interval. Resulting trajectories of model runs for each individual recovery unit, and the western North Atlantic population as a whole, were found to be very similar. The model run estimates from the adult female population size for the western North Atlantic (from the 2004-2008 time frame), suggest the adult female population size is approximately 20,000-40,000 individuals, with a low likelihood of females' numbering up to 70,000 (NMFS 2009a). A less robust estimate for total benthic females in the western North Atlantic was also obtained, yielding approximately 30,000-300,000 individuals, up to less than 1 million (NMFS 2009a). A preliminary regional abundance survey of loggerheads within the northwestern Atlantic continental shelf for positively identified loggerhead in all strata estimated about 588,000 loggerheads (interquartile range of 382,000-817,000). When correcting for unidentified turtles in proportion to the ratio of identified turtles, the estimate increased to about 801,000 loggerheads (interquartile range of 521,000-1,111,000) (NMFS 2011c).

4.1.1.5.4 Threats (Specific to Loggerhead Sea Turtles)

The threats faced by loggerhead sea turtles are well summarized in the general discussion of threats in Section 4.1.1.1 of this Opinion. Yet the impact of fishery interactions is a point of further emphasis for this species. The joint NMFS and USFWS Loggerhead Biological Review Team determined that the greatest threats to the NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant et al. 2009).

Regarding the impacts of pollution, loggerheads may be particularly affected by organochlorine contaminants; they have the highest organochlorine concentrations (Storelli et al. 2008) and metal loads (D'Ilio et al. 2011) in sampled tissues among the sea turtle species. It is thought that dietary preferences were likely to be the main differentiating factor among sea turtle species. Storelli et al. (2008) analyzed tissues from stranded loggerhead sea turtles and found that mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991).

While oil spill impacts are discussed generally for all species in Section 4.1.1.1 of this Opinion, specific impacts of the DWH oil spill event on loggerhead sea turtles are considered here. Impacts to loggerhead sea turtles occurred to offshore small juveniles as well as large juveniles and adults. A total of 30,800 small juvenile loggerheads (7.3% of the total small juvenile sea turtle exposures to oil from the spill) were estimated to have been exposed to oil. Of those exposed, 10,700 small juveniles are estimated to have died as a result of the exposure. In contrast to small juveniles, loggerheads represented a large proportion of the adults and large juveniles exposed to and killed by the oil. There were 30,000 exposures (almost 52% of all exposures for those age/size classes) and 3,600 estimated mortalities. A total of 265 nests (27,618 eggs) were also translocated during response efforts, with 14,216 hatchlings released, the fate of which is unknown (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Additional unquantified effects may have included inhalation of volatile compounds, disruption of foraging or migratory movements due to surface or subsurface oil, ingestion of prey species contaminated with oil and/or dispersants, and loss of foraging resources which could lead to compromised growth and/or reproductive potential. There is no information currently available to determine the extent of those impacts, if they occurred.

Unlike Kemp's ridleys, the majority of nesting for the Northwest Atlantic Ocean loggerhead DPS occurs on the Atlantic coast, and thus loggerheads were impacted to a relatively lesser degree. However, it is likely that impacts to the Northern Gulf of Mexico recovery unit of the NWA loggerhead DPS would be proportionally much greater than the impacts occurring to other recovery units. Impacts to nesting and oiling effects on a large proportion of the Northern Gulf of Mexico recovery unit recovery unit, especially mating and nesting adults likely had an impact on the Northern Gulf of Mexico recovery unit. Based on the response injury evaluations for Florida Panhandle and Alabama nesting beaches (which fall under the Northern Gulf of Mexico recovery unit), the Trustees estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH oil spill response activities on nesting beaches. Although the long-term effects remain unknown, the DWH oil spill event impacts to the Northern Gulf of Mexico Recovery Unit may result in some nesting declines in the future due to a large reduction of oceanic age classes during the DWH oil spill event. Although adverse impacts occurred to loggerheads, the proportion of the population that is expected to have been exposed to and directly impacted by the DWH oil spill event is relatively low. Thus we do not believe a population-level impact occurred due to the widespread distribution and nesting location outside of the Gulf of Mexico for this species.

Specific information regarding potential climate change impacts on loggerheads is also available. Modeling suggests an increase of 2°C in air temperature would result in a sex ratio of over 80% female offspring for loggerheads nesting near Southport, North Carolina. The same increase in air temperatures at nesting beaches in Cape Canaveral, Florida, would result in close to 100% female offspring. Such highly skewed sex ratios could undermine the reproductive capacity of the species. More ominously, an air temperature increase of 3°C is likely to exceed the thermal threshold of most nests, leading to egg mortality (Hawkes et al. 2007). Warmer sea surface temperatures have also been correlated with an earlier onset of loggerhead nesting in the spring (Hawkes et al. 2007; Weishampel et al. 2004), short inter-nesting intervals (Hays et al. 2002), and shorter nesting seasons (Pike et al. 2006).

4.1.2 Atlantic Sturgeon

Five separate DPSs of Atlantic sturgeon were listed under the ESA by NMFS effective April 6, 2012 (77 FR 5880 and 5914, Publication Date February 6, 2012). The New York Bight, Chesapeake Bay, Carolina, and SA DPSs were listed as endangered. The Gulf of Maine DPS was listed as threatened. Because adult and subadult Atlantic sturgeon from all DPSs mix extensively in marine waters, we expect fish from all DPSs to be found in the action area.

4.1.2.1 Species Descriptions and Distributions

Atlantic sturgeon are long-lived, late-maturing, estuarine-dependent, anadromous fish distributed along the eastern coast of North America (Waldman and Wirgin 1998). Historically, sightings have been reported from Hamilton Inlet, Labrador, Canada, south to the St. Johns River, Florida (Murawski and Pacheco 1977; Smith and Clugston 1997). Atlantic sturgeon may live up to 60 years, reach lengths up to 14 ft, and weigh over 800 lbs (Atlantic Sturgeon Status Review Team 2007; Collette and Klein-MacPhee (editors) 2002). They are distinguished by armor-like plates (called scutes) and a long protruding snout that has 4 barbels (slender, whisker-like feelers extending from the lower jaw used for touch and taste). Adult Atlantic sturgeon spend the majority of their lives in nearshore marine waters, returning to the rivers where they were born (natal rivers) to spawn (Wirgin et al. 2002). Young sturgeon may spend the first few years of life in their natal river estuary before moving out to sea (Wirgin et al. 2002). Atlantic sturgeon are omnivorous benthic (bottom) feeders and incidentally ingest mud along with their prey. Diets of adult and subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Atlantic Sturgeon Status Review Team 2007; Bigelow and Schroeder 1953a; Guilbard et al. 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Atlantic Sturgeon Status Review Team 2007; Bigelow and Schroeder 1953a; Guilbard et al. 2007).

Historically, Atlantic sturgeon were present in approximately 38 rivers in the United States from the St. Croix River, Maine to the St. Johns River, Florida, of which 35 rivers have been confirmed to have had a historical spawning subpopulation.

Presently, the SA DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) of the Ashepoo, Combahee, and Edisto Rivers' basin southward along the South Carolina, Georgia, and Florida coastal areas to the Saint Johns River, Florida.

The Carolina DPS includes all Atlantic sturgeon that are spawned in the watersheds (including all rivers and tributaries) from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor.

The Chesapeake Bay DPS is comprised of Atlantic sturgeon that originate from rivers that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, Virginia.

The New York Bight DPS includes all anadromous Atlantic sturgeon that spawn in the watersheds that drain into coastal waters from Chatham, Massachusetts, to the Delaware-Maryland border on Fenwick Island.

The Gulf of Maine DPS includes all anadromous Atlantic sturgeon that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, Massachusetts.

The marine range of all 5 DPS of Atlantic sturgeon extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida. The action area includes waters (marine, estuarine, and rivers) from the Virginia/North Carolina border south to Florida.

4.1.2.2 Life History Information (Applicable to all DPSs)

Atlantic sturgeon are generally referred to as having 4 size/developmental categories: larvae; young-of-year; juveniles and subadults; and adults. Hatching occurs approximately 94-140 hours after egg deposition. Immediately after hatching larvae enter the yolk sac larval stage and assume a demersal existence (Smith et al. 1980). The yolk sac provides nutrients to the animals until it is completely absorbed 8-12 days after hatching (Kynard and Horgan 2002). Animals in this stage are less than 4 weeks old, with total lengths less than 30 mm (van Eenennaam et al.

1996). Animals in this phase are in freshwater and are located far upstream very near the spawning beds. As the larvae develop they commence downstream migration towards the estuaries. During the first half of their downstream migration, movement is limited to night. During the day, larvae use gravel, rocks, sticks, etc. as refugia (Kynard and Horgan 2002). During the latter half of migration when larvae are more fully developed, movement occurs both day and night. Salinities of 5-10 parts per thousand are known to cause mortality at this young stage (Bain 1997; Cech Jr. and Doroshov 2005; Kynard and Horgan 2002).

As larvae grow and absorb the yolk sac, they enter the young-of-year phase. Young-of-year are greater than 4 weeks old but less than 1 year, and generally occur in the natal river. These animals are generally located downstream of the spawning beds in primarily freshwater, though they can be found in the estuaries.

Following the young-of-year life phase, sturgeon develop into juveniles and subadults. There is little morphometric difference, aside from overall size, between juveniles and subadults; they are primarily distinguished by their occurrence within estuarine or marine waters. Juveniles are generally only found in estuarine habitats, while subadults may be found in estuarine and marine waters. As a group, juveniles and subadults range in size from approximately 300-1500 mm total length. The term "juveniles" refers to animals 1 year of age or older that reside in the natal estuary. Estuarine habitats are important for juveniles, serving as nursery areas by providing abundant foraging opportunities, as well as thermal and salinity refuges, for facilitating rapid growth. During their first 2 years, juvenile Atlantic sturgeon remain in the estuaries of their natal rivers, which may include both fresh and brackish channel habitats below the head of tide (Hatin et al. 2007). Upon reaching age 2, juveniles become increasingly salt tolerant and some individuals will begin their outmigration to nearshore marine waters (Bain 1997; Dovel and Berggren 1983; Hatin et al. 2007). Some juveniles will take up residency in non-natal rivers that lack active spawning sites (Bain 1997). By age 5, most juveniles have completed their transition to saltwater becoming "subadults," "late-stage juveniles," or "marine migratory juveniles"; however, these animals are frequently encountered in estuaries of non-natal rivers (Bahr and Peterson 2016).

Out migration of larger juveniles may be influenced by the density of younger, less-developed juveniles. Because early juveniles are intolerant of salinity, they are likely unable to use foraging habitats in coastal waters if riverine food resources become limited. However, older, more-developed juveniles are able to use these coastal habitat, though they may prefer the relatively predator-free environments of brackish water estuaries as long as food resources are not limited (Schueller and Peterson 2010).

Adults are sexually mature individuals of 1500+ mm total length and 5 years of age or older. They may be found in freshwater riverine habitats on the spawning grounds or making migrations to and from the spawning grounds. They also use estuarine waters seasonally, principally in the spring through fall and will range widely in marine waters during the winter. After emigration from the natal estuary, subadults and adults travel within the marine environment, typically in waters less than 50 m in depth, using coastal bays, sounds, and ocean waters often occurring over sand and gravel substrate (Bigelow et al. 1963; Collins and Smith 1997; Dovel and Berggren 1983; Dunton et al. 2010; Erickson et al. 2011; Greene et al. 2009;

Laney et al. 2007; Murawski and Pacheco 1977; Savoy and Pacileo 2003; Smith 1985; Stein et al. 2004b; Welsh et al. 2002; Wirgin and King 2011).

Atlantic sturgeon populations show clinal variation, with a general trend of faster growth and earlier age at maturity in more southern systems. Atlantic sturgeon mature between the ages of 5 and 19 years in South Carolina (Smith et al. 1982), between 11 and 21 years in the Hudson River (Young et al. 1988), and between 22 and 34 years in the St. Lawrence River (Scott and Crossman 1973). Atlantic sturgeon likely do not spawn every year. Multiple studies have shown that spawning intervals range from 1 to 5 years for males (Caron et al. 2002; Collins et al. 2000b; Smith 1985) and 2 to 5 years for females (Bigelow et al. 1963; Stevenson and Secor 1999; van Eenennaam et al. 1996). Fecundity (number of eggs) of Atlantic sturgeon has been correlated with age and body size, with egg production ranging from 400,000 to 8,000,000 eggs per female per year (Dadswell 2006; Smith et al. 1982; van Eenennaam and Doroshov 1998). The average age at which 50% of maximum lifetime egg production is achieved is estimated to be 29 years, approximately 3 to 10 times longer than for other bony fish species examined (Boreman 1997).

Spawning adult Atlantic sturgeon generally migrate upriver in spring to early summer, which occurs in February-March in southern systems, April-May in Mid-Atlantic systems, and May-July in Canadian systems (Bain 1997; Caron et al. 2002; Murawski and Pacheco 1977; Smith 1985; Smith and Clugston 1997). Likely fall spawning runs have been identified in the Edisto River, South Carolina (Farrae et al. 2017b) and the Altamaha River, Georgia (Ingram and Peterson 2016). Telemetry data collected in 2013 and 2015 also show acoustically tagged fish making spawning runs in late summer (August – September) in the Savannah River (South Carolina Department of Natural Resources, Unpublished data). A fall spawning run has also been confirmed in the Roanoke River, North Carolina (Smith et al. 2014), in the Carolina DPS; however, they report a spring spawning run is also likely occurring. This suggests that a fall spawn is occurring in a number of southern rivers (Collins et al. 2000b; McCord et al. 2007; Moser et al. 1998; Rogers and Weber 1995; Weber and Jennings 1996). Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers, when and where optimal flows are 46-76 centimeters per second and depths are 3-27 m (Bain et al. 2000; Borodin 1925; Crance 1987; Leland III 1968; Scott and Crossman 1973). Males commence upstream migration to the spawning sites when waters reach around 6°C (Dovel and Berggren 1983; Smith 1985; Smith et al. 1982) with females following a few weeks later when water temperatures are closer to 12° or 13°C (Collins et al. 2000a; Dovel and Berggren 1983; Smith 1985). Atlantic sturgeon have highly adhesive eggs that must be laid on hardbottom in order to stick. Thus, spawning occurs over hard substrate, such as cobble, gravel, or boulders (Gilbert 1989a; Smith and Clugston 1997).

4.1.2.3 Status and Population Dynamics

At the time Atlantic sturgeon were listed, the best available abundance information for each of the 5 DPSs was the estimated number of adult Atlantic sturgeon spawning in each of the rivers on an annual basis (77 FR 5914; Publication Date February 6, 2012). However, the estimated number of annually spawning adults in each of the river subpopulations is insufficient to quantify the total population numbers for each DPS of Atlantic sturgeon due to the lack of other necessary accompanying life history data. In 2012, the NEFSC estimated the total ocean population of adults and subadults, vulnerable to capture in fisheries within the sampling domain of the

Northeast Area Monitoring and Assessment Program (NEAMAP). NEAMAP trawl surveys were conducted from Cape Cod, Massachusetts, to Cape Hatteras, North Carolina, in nearshore waters to depths of 60 ft from fall 2007 through spring 2012.

The results of these surveys are presented in Table 30. It is important to note that the NEAMAP surveys were conducted primarily in the Northeast and may underestimate the actual population abundances of the Carolina and South Atlantic DPSs, which are likely more concentrated in the Southeast since they originated from and spawn there. However, the total ocean population abundance estimates listed in Table 30 currently represent the best available population abundance estimates for the 5 U.S. Atlantic sturgeon DPSs.

Table 30. Summary of Calculated Population Estimates based upon the NEAMAP Survey Swept Area, Assuming 50% Efficiency

DPS	Mean Percent Composition Estimate for Each DPS (Damon-Randall et al. 2013)[45]	Estimated Ocean Population Abundance (NMFS 2013)	Estimated Ocean Population of Adults (NMFS 2013)	Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries) (NMFS 2013)
South Atlantic	20%	14,911	3,728	11,183
Carolina	4%	1,356	339	1,017
Chesapeake Bay	14%	8,811	2,203	6,608
New York Bight	49%	34,566	8,642	25,925
Gulf of Maine	11%	7,455	1,864	5,591

Since the 2012 listing, 2 mixed stock analyses have been completed for Atlantic sturgeon – 1 evaluating individuals captured in the Northeast United States (from north of Cape Hatteras to Maine) and 1 evaluating individuals in the Southeast United States (from south of Cape Hatteras to central east Florida). A mixed stock analysis is currently the best available method for identifying which DPSs are most likely to be encountered in the marine environment. A mixed stock analysis pools all the genetic information available for Atlantic sturgeon caught in a given area and evaluates the proportional representation of each river of origin and DPS within that catch area. The proportion of animals from a specific DPS or river of origin found in a given catch area, is directly related to the distance between the catch area and those rivers/DPSs. For example, the South Atlantic DPS includes rivers from Florida, Georgia, and South Carolina. Thus, for a catch area off the coast of Georgia, we anticipate a high proportion of individuals from the South Atlantic DPS and a low proportion from the Gulf of Maine DPS. Similarly, individuals from DPSs with larger populations are expected to occur at higher proportions overall than animals from DPSs with relatively smaller populations.

NMFS Greater Atlantic Region applied the results of the mixed stock analysis specific to the Northeast Region (Damon-Randall et al. 2013) to their 2012 estimate (Table 30), to estimate the

[45] Mean percent composition for animals from the St. John, Canada, population are not considered here because they were not listed under the ESA. Thus, these percentages do not add to 100%.

likely population of adults and subadults from each DPS captured in the NEAMAP trawl surveys conducted from Cape Cod, Massachusetts, to Cape Hatteras, North Carolina.

The U.S. Geological Service-Leetown Science Center completed a draft mixed stock analysis specific to the Southeast Region in late 2019 (USGS unpublished data). USGS provided information on both river of origin and DPS of origin (Table 31). Specifically, the report evaluated the genetic information from a given individual and determined which of 14 potential reference rivers it was most likely to have originated from. Individuals found in Southeast only assigned to 12 of those reference rivers; individuals from 2 rivers in Canada (St. John and St. Lawrence) were not detected, as shown in Table 31. USGS (unpublished data) used the same approach to assign individuals in the Southeast to a likely DPS of origin.

Prior to the completion of the Southeast-specific mixed stock analysis, we used the population estimates (Table 30), and mixed stock analysis for the Northeast (Damon-Randall et al. 2013), for projects occurring in the Southeast because they represented the best available information. This Southeast-specific mixed stock analysis significantly improves the accuracy with which we can assign incidental takes occurring for federal actions taking place in the Southeast. However, with the new analysis, it is no longer appropriate to use the total ocean population estimates of adults and subadults based on data from the NEAMAP program, because those estimates were based on individuals collected outside the Southeast. Unfortunately, no estimates of the total populations in the Southeast have been completed. In the absence of a total population estimate, we relied on the best river-specific estimates available to develop an estimate of abundance for Atlantic sturgeon in the Southeast. We therefore calculated population estimates of the juvenile and adult populations of Atlantic sturgeon in the Southeast (described below) using demographic information from the Altamaha River combined with the proportions on individuals from specific rivers and DPSs provided in the draft 2019 mixed stock analysis (USGS unpublished data).

We used the Altamaha River in the South Atlantic DPS as the foundation for our estimates because it has the most available information. The scientific literature provides estimates of Age 1, 2, and 3 abundances for the Altamaha River in 2004-2006 (Schueller and Peterson 2010), as well as estimates of the number of adults likely making spring spawning runs in 2004 and 2005 (Peterson et al. 2008). Ingraham and Peterson (2016) calculated the likely proportion of sexually mature adults in the Altamaha system that make spawning runs in the spring, allowing us to extrapolate the total number of spawning adults in the Altamaha River. We summed the estimates for all age classes (i.e., Age 1 juveniles to extrapolated total spawning adults) to estimate a likely total population of Atlantic sturgeon in the Altamaha River. Following this approach, we estimated the minimum total juvenile and adult spawning population in the Altamaha River was between 1,940 and 2,525 individuals (Table 31).

Once we estimated the likely total population for the Altamaha River for 2004 and 2005, we used the information in the draft 2019 mixed stock analysis (USGS unpublished data) to calculate the likely number of individuals from other rivers/DPS that likely occur in the Southeast. Specifically, since we estimated the minimum total juvenile and adult spawning population in the Altamaha River was between 1,940 and 2,525 individuals, and the mixed stock analysis estimated the Altamaha River accounted from approximately 20.2% of the individuals in the Southeast, we estimated the total Southeast population of Atlantic sturgeon as between 9,583-12,477 (5,867-27,387). Table 31 outlines this approach and provides our estimates.

The resulting estimates are conservative and likely represent a minimum numbers of animals because they do not account for young-of-year individuals. Young-of-year individuals are so small they are difficult to capture and were not a focus of the Altamaha River sampling. Likewise, they only estimate the likely population present in the Southeast (e.g., south of Cape Hatteras to central east Florida). This is significant because while individuals from the northern DPSs (Gulf of Maine, New York Bight, and Chesapeake Bay DPSs) could be in the action area, those animals are likely transients.. Thus, our estimates of individuals from those DPSs occurring in the Southeast is unlikely to accurately reflect the total population abundance for the Gulf of Maine, New York Bight, and Chesapeake Bay DPSs. Additionally, some portion of the South Atlantic and Carolina DPSs are likely to travel north of Cape Hatteras. We discuss available information and population estimates by DPS below.

Table 31. Estimated Atlantic Sturgeon Population in the Southeast[46]

Step 1				
Year	Age 1 (Min/Max)	Age2 (Min/Max)	Age 3 (Min/Max)	Total (Min/Max)
2004	483 (368-643)	544 (424-707)	37 (9-294)	1,064 (801-1,644)
2005	1345 (1,077-1,697)	107 (28-784)	30 (6-935)	1,482 (1,111-3,416)
Step 2				
Year	Estimated Spring Spawning Run (Min/Max)	Estimated Proportion of Adults Making Spring Spawning Run (Min/Max)	Total Adults (Min/Max)	
2004	324 (143-667)	0.37 (0.36-0.38)	876 (386-1,802)	
2005	386 (216-787)	0.37 (0.36-0.38)	1,043 (584-2,217)	
Step 3				
Year	Total (Min/Max)			
2004	1,940 (1,187-3,447)			
2005	2,525 (1,695-5,542)			
Step 4				
River Population (DPS)	Proportion of Individuals from River Population (CI)	DPS	Proportion of Individuals from DPS (CI)	
Kennebec (Gulf of Maine)	0.001 (0-0.004)	Gulf of Maine	0.001(0-0.004)	
Hudson (New York Bight)	0.025 (0.015-0.035)	New York Bight	0.036 (0.025-0.048)	
Delaware (New York Bight)	0.011 (0.004-0.0200	Chesapeake	0.096 (0.079-0.121)	
York (Chesapeake Bay)	0.004 (0.001-0.008)	Carolina	0.338 (0.292-0.364)	
James (Chesapeake Bay)	0.093 (0.076-0.116)	South Atlantic	0.529 (0.499-0.570)	
Albemarle Complex (Carolina)	0.309 (0.259-0.331)			
Pee Dee (Carolina)	0.030 (0.011-0.053)			
Edisto Spring (South Atlantic)	0.020 (0.012-0.034)			
Edisto Fall (South Atlantic)	0.100 (0.081-0.124)			

[46] Follows these steps: Step 1 – Sum Juvenile Abundance in Altamaha River (Schueller and Peterson 2010); Step 2 – Estimate Adult Population in Altamaha River (Ingram and Peterson 2016; Peterson et al. 2008a), Step 3 – Sum Age Class Estimates in Altamaha River, Step 4 – Identify the Proportion of Individual River Systems/DPS Represented in the Southeast (USGS unpublished data), and Step 5 – Estimate Individuals from Remaining River Populations and DPSs, using Altamaha Estimate.

Step 4				
Savannah (South Atlantic)	0.102 (0.070-0.137)			
Ogeechee (South Atlantic)	0.070 (0.054-0.098)			
Altamaha (South Atlantic)	0.202 (0.171-0.238)			
Satilla (South Atlantic)	0.036 (0.020-0.049)			

Step 5				
River Population (DPS)	Proportion of Individuals in SE by River Population (CI)	Minimum Number of Individuals in SE by River Populations (Min/Max)	Proportion of Individuals from DPS	Minimum Number of Individuals in SE By DPS
Kennebec (Gulf of Maine)	0.001 (0-0.004)	9 (5-25)	Gulf of Maine – 0.001 (0-0.004)	9
Hudson (New York Bight)	0.025 (0.015-0.035)	241 (147-687)	New York Bight – 0.036 (0.025-0.048)	343
Delaware (New York Bight)	0.011 (0.004-0.0200	103 (63-293)	Chesapeake – 0.096 (0.079-0.121)	920
York (Chesapeake Bay)	0.004 (0.001-0.008)	35 (21-99)	Carolina – 0.338 (0.292-0.364)	3,243
James (Chesapeake Bay)	0.093 (0.076-0.116)	886 (543-2,533)	South Atlantic – 0.529 (0.499-0.570)	5,067
Albemarle Complex (Carolina)	0.309 (0.259-0.331)	2,957 (1,810-8,451)		
Pee Dee (Carolina)	0.030 (0.011-0.053)	286 (175-816)		
Edisto Spring (South Atlantic)	0.020 (0.012-0.034)	188 (115-537)		
Edisto Fall (South Atlantic)	0.100 (0.081-0.124)	954 (584-2,725)		
Savannah (South Atlantic)	0.102 (0.070-0.137)	973 (596-2,780)		
Ogeechee (South Atlantic)	0.070 (0.054-0.098)	666 (408-1,903)		
Altamaha (South Atlantic)	0.202 (0.171-0.238)	1,940 (1,187-5,543)		
Satilla (South Atlantic)	0.036 (0.020-0.049)	348 (213-994)		

SA DPS

Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in Georgia prior to 1890.

The SA DPS historically supported 8 spawning subpopulations. At the time of listing only the following 6 spawning subpopulations were believed to have existed: Combahee River, Edisto River, Savannah River, Ogeechee River, Altamaha River (including the Oconee and Ocmulgee tributaries), and Satilla River. We determined those rivers/river systems supported spawning if YOY were observed or mature adults were present in freshwater portions of a system. Three of the spawning subpopulations in the South Atlantic DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. Peterson et al. (2008) estimated the number of spawning adults in the Altamaha River was 324 (95% CI: 143-667) in 2004 and 386 (95% CI: 216-787) in 2005. The Altamaha and Combahee/Edisto River spawning subpopulations are likely less than 6% of their historical abundance. For the remaining spawning rivers, less than 300 adults are estimated to be spawning annually (total of both sexes) (75 FR 61904; October 6, 2010). Bahr and Peterson (2016) estimated the age-1 juvenile abundance in the Savannah River from 2013-2015 at 528 in 2013, 589 in 2014, and 597 in 2015. The abundance of the remaining 3 spawning subpopulations in the South Atlantic DPS is likely less than 1% of their historical abundance (ASSRT 2007).

The 2 remaining historical spawning subpopulations in the Broad-Coosawatchie River and St. Marys River were believed to be extinct. However, new information provided from the capture of juvenile Atlantic sturgeon suggests the spawning subpopulation in the St. Marys River is not extinct and continues to exist, albeit at very low levels. Regardless of river, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development.

In 2017, the Atlantic States Marine Fisheries Commission (ASMFC) completed an Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017; Atlantic States Marine Fisheries Commission 2017). The purpose of the assessment was to evaluate the status of Atlantic sturgeon along the U.S. Atlantic coast (ASMFC 2017). The assessment considered the status of each DPS individually, as well as all 5 DPSs collectively as a single unit. The assessment determined the SA DPS abundance is "depleted" relative to historical levels. The assessment concluded there was not enough information available to assess the abundance of the DPS since the implementation of the 1998 fishing moratorium. However, it did conclude there was 40% probability the SA DPS is still subjected to mortality levels higher than determined acceptable in the 2017 assessment.

The assessment also estimated effective population sizes (N_e) when possible. Effective population size is generally considered to be the number of individuals that contribute offspring to the next generation. More specifically, based on genetic differences between animals in a given year, or over a given period of time, researchers can estimate the number of adults needed to produce that level of genetic diversity. For the South Atlantic DPS, the assessment reported N_e for the Edisto, Savannah, Ogeechee, and Altamaha rivers (Table 32). Additional estimates of N_e have been conducted since the completion of the assessment, including for additional river systems; Table 32 reports those estimates.[47]

Table 32. Estimates of Effective Population Size by Rivers

River	Effective Population Size (N_e) (95% CI)	Sample Size	Collection Years	Reference
Edisto	55.4 (36.8-90.6)	109	1996-2005	ASMFC (2017)
	Fall Run – 48.0 (44.7-51.5)	1,154	1996-2004	Farrae et al. (2017a)
	Spring Run – 13.3 (12.1-14.6)	198	1998, 2003	Farrae et al. (2017a)
	60.0 (51.9-69.0)	145	1996, 1998, 2005	Waldman et al. (2018)
Savannah	126.5 (88.1-205)	98	2000-2013	ASMFC (2017)
	123 (103.1-149.4)	161	2013, 2014, 2017	Waldman et al. (2018)
Ogeechee	32.2 (26.9-38.8)	115	2003-2015	ASMFC (2017)

[47] The effective population size estimates in Table 32 are different from the estimated population estimated Atlantic sturgeon population in the Southeast reported in the Table 31. The effective population size estimates refer to the likely number of unique spawning individuals needed to produce the level of genetic variability seen in the population. The effective population size only considers spawning individuals and does not account for any other age classes. Therefore, an estimate of effective population size is *not* an estimate of population abundance and can be significantly lower than the total number of individuals when account for all age classes.

River	Effective Population Size (N_e) (95% CI)	Sample Size	Collection Years	Reference
	26 23.9–28.2	200	2007-2009, 2014-2017	Waldman et al. (2018)
	23.9 (22.2-25.7)	197	2007-2009, 2014-2017	Fox et al. (2019)
Altamaha	111.9 (67.5-216.3)	186	2005-2015	ASMFC (2017)
	149 (128.7–174.3)	245	2005, 2011, 2014, 2016-2017	Waldman et al. (2018)
	142.1 (124.2-164.0)	268	2005, 2011, 2014-2017	Fox et al. (2019)
Satilla	21 (18.7–23.2)	68	2015-2016	Waldman et al. (2018)
St. Marys	1 (1.3–2.0)	14	2014-2015	Waldman et al. (2018)

Generally, a minimum N_e of 100 individuals is considered the threshold required to limit the loss in total fitness from in-breeding depression to <10%; while an N_e greater than 1,000 is the recommended minimum to maintain evolutionary potential (ASMFC 2017; Frankham et al. 2014). N_e is useful for defining abundance levels where populations are at risk of loss of genetic fitness (ASMFC 2017). While not inclusive of all the spawning rivers in the South Atlantic DPS, the estimates reported in Table 32 suggest there is a risk for inbreeding depression (N_e < 100) in 4 of those rivers (Edisto, Ogeechee, Satilla, and St. Marys rivers) and loss of evolutionary potential (N_e < 1000) in all six. This information suggests there at least some inbreeding depression within the DPS and loss of evolutionary potential throughout all of it. The NMFS Greater Atlantic Region NEAMAP model estimates a minimum ocean population for the entire South Atlantic DPS of 14,911 Atlantic sturgeon, of which 3,728 are adults. The SERO estimate, based on the 2019 USGS mixed stock analysis, is that the minimum number of individuals from the SA DPS occurring in the Southeast is 5,067.

Carolina DPS
Historical fishery landings data indicate between 7,000 and 10,500 adult female Atlantic sturgeon were present in North Carolina prior to 1890 (Armstrong and Hightower 2002; Secor 2002). Secor (2002) estimates that 8,000 adult females were present in South Carolina during that same time-frame. At the time of listing, the abundance for each river population within the DPS was estimated to have fewer than 300 spawning adults; estimated to be less than 3% of what they were historically (Atlantic Sturgeon Status Review Team 2007).

NMFS identified 7 rivers/river systems within the Carolina DPS where spawning is likely occurring (Roanoke; Tar- Pamlico; Neuse; Cape Fear and Northeast Cape Fear; Pee Dee, Waccamaw, Bull Creek; Black; Santee, and Cooper). We determined those rivers/river systems supported spawning if young-of-year were observed, or mature adults were present, in freshwater portions of a system. However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development.

Historically, both the Sampit and Ashley Rivers in South Carolina were documented to have spawning subpopulations at one time. Yet, the spawning subpopulation in the Sampit River is believed to be extirpated and the current status of the spawning subpopulation in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning subpopulations.

The Assessment determined the Carolina DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (67%) that the Carolina DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a relatively high probability (75%) the Carolina DPS is still subjected to mortality levels higher than determined acceptable in the 2017 Assessment (Atlantic States Marine Fisheries Commission 2017).

For the Carolina DPS, the Assessment only reported N_e for the Albemarle Sound. Based on samples collected from 37 individuals from 1998-2008, the Assessment estimated an N_e of 14.2 individuals (Atlantic States Marine Fisheries Commission 2017). While not inclusive of all the spawning rivers in the Carolina DPS, this estimate suggests there is a risk for both inbreeding depression ($N_e < 100$) and loss of evolutionary potential ($N_e < 1000$) in the DPS, assuming Albemarle Sound is representative of the entire DPS. The NMFS Greater Atlantic Region NEAMAP model estimates a minimum ocean population for the entire Carolina DPS of 1,356 Atlantic sturgeon, of which 339 are adult. We estimate the minimum number of individuals from the Carolina DPS occurring in the Southeast is 3,243.

Chesapeake Bay DPS
Historically, the Chesapeake Bay DPS likely supported more than 10,000 spawning adults (Atlantic Sturgeon Status Review Team 2007; Maine State Planning Office 1993; Secor 2002). Currently, there are 4 known spawning subpopulations for the Chesapeake Bay DPS, one each for the Pamunkey River and for Marshyhope Creek, and 2 for the James River (Balazik et al. 2017; Balazik et al. 2012a) (Balazik and Musick 2015; Greenlee and Secor 2016; Hager et al. 2014; Richardson and Secor 2017). Atlantic sturgeon that are spawned elsewhere are known to use waters of the Chesapeake Bay for other life functions, such as foraging and as juvenile nursery habitat, before entering the marine system as subadults (Atlantic Sturgeon Status Review Team 2007; Bigelow et al. 1963; Grunwald et al. 2008; Wirgin et al. 2007).

The existence of the Pamunkey River spawning subpopulation was identified in 2013 after the capture of spawning condition adults (e.g., males expressing milt, and females with eggs) within tidal freshwater of the river during the late summer to early fall (i.e., August - October) (Hager et al. 2014). Based on the capture of 17 sturgeon, Kahn et al. (2014) estimated 75 adults (95% confidence interval = 17–168 adults) spawned in the river in 2013. There are no other estimates of abundance for this spawning subpopulation or trends in abundance.

The Marshyhope Creek spawning subpopulation was identified in 2014, likewise after the capture of spawning condition adults during the late summer to early fall. Twenty-six adults, including males expressing milt and females with ripe eggs, have been captured in Marshyhope Creek since 2014. DNA analysis is ongoing to determine whether the sturgeon are part of a naturally occurring population or are hatchery fish that were released into the Nanticoke River in 1996 (Greenlee and Secor 2016; Richardson and Secor 2017; Secor et al. 2000). There are no estimates of abundance or trends in abundance for this spawning subpopulation.

At the time of listing, the James River was the only known spawning river for the Chesapeake Bay DPS and spawning was believed to occur only in the spring, from approximately April – May, based on historical and current evidence (Atlantic Sturgeon Status Review Team 2007).

Subsequently, new information for when and where spawning-condition adults were captured and tracked in the river led to the conclusion that Atlantic sturgeon spawn in the James River in both the spring and in the late summer to early fall (Balazik et al. 2012a; Balazik and Musick 2015). The results of genetic analyses support that the adults are 2 separate spawning groups. The genetic analyses also informed the effective population size of each group which were similar (Fall: N_e = 46 [95% CI: 32±71], Spring: N_e = 44 [95% CI: 26±79]) despite differences in the number of adults captured from each spawning subpopulation. From 2007 to 2016, 507 individual fall run Atlantic sturgeon were captured during the fall spawning and 40 individual Atlantic sturgeon were captured during the spring spawning (Balazik et al. 2017). This is a minimum count of the number of adult Atlantic sturgeon in the James River during the time period because capture efforts did not occur in all areas and at all times when Atlantic sturgeon were present in the river. There are no other estimates of abundance or trends in abundance for the James River spawning subpopulations.

The Assessment determined the Chesapeake Bay DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively low probability (37%) that the Chesapeake Bay DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 30% probability the Chesapeake Bay DPS is still subjected to mortality levels higher than determined acceptable in the 2017 Assessment.

The Assessment reported N_e for the York and James rivers in the Chesapeake Bay DPS. In the York River, samples from 136 individuals collected from 2013-2015 produced an estimated N_e of 7.8 individuals (Atlantic States Marine Fisheries Commission 2017). In the James River, 346 samples were collected from 1998-2015 and produced an estimated N_e of 40.9 individuals (Atlantic States Marine Fisheries Commission 2017). While not inclusive of all the spawning rivers in the Chesapeake Bay DPS, these estimates at least hint that there is a risk for both inbreeding depression ($N_e < 100$) and loss of evolutionary potential ($N_e < 1000$) in the DPS. The NMFS Greater Atlantic Region NEAMAP model estimates a minimum ocean population for the entire DPS of 8,811 Atlantic sturgeon, of which 2,319 are adults. We estimate the minimum number of individuals from the Chesapeake Bay DPS occurring in the Southeast is 920. Given the distance between the rivers of this DPS and Southeast, we anticipate these individuals would be sub-adults or adults.

New York Bight DPS
New York Bight DPS Atlantic sturgeon historically spawned in the Connecticut, Delaware, Hudson, and Taunton Rivers (Atlantic Sturgeon Status Review Team 2007; Murawski and Pacheco 1977; Secor 2002). Spawning still occurs in the Delaware and Hudson Rivers, and evidence of spawning was recently documented in the Connecticut River (Atlantic Sturgeon Status Review Team 2007) (Savoy et al. 2017). Atlantic sturgeon that are spawned elsewhere continue to use habitats within the Connecticut and Taunton Rivers for other life functions (Atlantic Sturgeon Status Review Team 2007; Savoy 2007; Wirgin and King 2011).

Prior to the onset of expanded fisheries exploitation of sturgeon in the 1800s, a conservative historical estimate for the Hudson River Atlantic sturgeon population was 10,000 adult females (Secor 2002). Current population abundance is likely at least one order of magnitude smaller than historical levels (Atlantic Sturgeon Status Review Team 2007; Kahnle et al. 2007; Secor

2002). Based on data collected from 1985-1995, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population (Kahnle et al. 2007). Kahnle (2007; 1998) also showed that the level of fishing mortality from the Hudson River Atlantic sturgeon fishery during the period of 1985-1995 exceeded the estimated sustainable level of fishing mortality for the riverine population, and may have led to reduced recruitment. At the time of listing, available data on abundance of juvenile Atlantic sturgeon in the Hudson River Estuary indicated a substantial drop in production of young since the mid-1970s (Kahnle et al. 1998). A decline appeared to occur in the mid- to late-1970s followed by a secondary drop in the late 1980s (Atlantic States Marine Fisheries Commission 2011; Kahnle et al. 1998; Sweka et al. 2007). CPUE data suggest that recruitment has remained depressed relative to catches of juvenile Atlantic sturgeon in the estuary during the mid- to late 1980s (Atlantic States Marine Fisheries Commission 2011; Sweka et al. 2007). From 1985-2007, there were significant fluctuations in CPUE. The number of juveniles appears to have declined between the late 1980s and early 1990s. While the CPUE is generally higher in the 2000s as compared to the 1990s, significant annual fluctuations make it difficult to discern any trend. The CPUEs from 2000-2007 are generally higher than those from 1990-1999; however, they remain lower than the CPUEs observed in the late 1980s. Standardized mean catch per net set from the New York State Department of Environmental Conservation juvenile Atlantic sturgeon survey have had a general increasing trend from 2006 – 2015, with the exception of a dip in 2013.There is currently not enough information regarding any life stage to establish a trend for the Hudson River population (Atlantic States Marine Fisheries Commission 2011; Sweka et al. 2007).

There is no abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800s indicate that this was historically a large population, with an estimated 180,000 adult females prior to 1890 (Secor 2002; Secor and Waldman 1999). Fisher (2009) sampled the Delaware River in 2009 to target young-of-year Atlantic sturgeon. The effort captured 34 young-of-year. Brundage and O'Herron (2003) also collected 32 young-of-year Atlantic sturgeon from the Delaware River in a separate study. Fisher (2011) reports that genetic information collected from 33 of the 2009 year class young-of-year indicates that at least 3 females successfully contributed to the 2009 year class. The capture of young-of-year in some years since 2009 shows that successful spawning is still occurring in the Delaware River. Based on the capture of juvenile Atlantic sturgeon in the Delaware River, researchers estimated estimate there were 3,656 (95% CI = 1,935–33,041) age 0-1 juvenile Atlantic sturgeon in the Delaware River subpopulation in 2014 (Hale et al. 2016). However, the relatively low numbers of captured adults suggest the existing riverine subpopulation is limited in size. For example, of the 261 adult-sized Atlantic sturgeon captured for scientific purposes off the Delaware Coast between 2009 and 2012, 100 were subsequently identified by genetics analysis to belong to the Hudson River subpopulation while only 36 belonged to the Delaware River subpopulation (Wirgin et al. 2015). Similar to the Hudson River, there is currently not enough information to determine a trend for the Delaware River population. The Atlantic Sturgeon Status Review Team (2007) suggested that there may be less than 300 spawning adults per year for the Delaware River portion of the New York Bight DPS.

The 2017 Assessment determined the New York Bight DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (75%) that the New

York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 31% probability the New York Bight DPS is still subjected to mortality levels higher than determined acceptable in the 2017 Assessment (Atlantic States Marine Fisheries Commission 2017).

The 2017 Assessment reported N_e for the Hudson and Delaware rivers in the New York Bight DPS. In the Hudson River, samples from 337 individuals collected from 1996-2015 produced an estimated N_e of 144.2 individuals (Atlantic States Marine Fisheries Commission 2017). In the Delaware River, 181 samples were collected from 2009-2015 and produced an estimated N_e of 56.7 individuals (Atlantic States Marine Fisheries Commission 2017). While not inclusive of all the spawning rivers in the New York Bight DPS, the estimates for the Hudson River suggests that spawning subpopulation may be large enough to avoid inbreeding depression ($N_e < 100$); however, the Delaware River spawning subpopulation may still be at risk. Both spawning subpopulations are likely at risk losing evolutionary potential ($N_e < 1000$). The NMFS Greater Atlantic Region NEAMAP model estimates a minimum ocean population for the entire DPS of 34,566 Atlantic sturgeon, of which 8,642 are adults. We estimate the minimum number of individuals from the New York Bight DPS occurring in the Southeast is 343. Given the significant distance between the rivers of this DPS and Southeast, we anticipate these individuals would be adults.

Gulf of Maine DPS

Gulf of Maine DPS Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (Atlantic Sturgeon Status Review Team 2007). Spawning still occurs in the Kennebec River, and captures of adult Atlantic sturgeon in the Androscoggin River, including a ripe male, over suitable spawning grounds during the spawning season confirm likely spawning; however, Atlantic sturgeon eggs and larvae have not yet been recovered in the Androscoggin (Wippelhauser pers. comm. 2018). The movement of subadult and adult sturgeon between rivers, including to and from the Kennebec River and the Penobscot River, demonstrates that coastal and marine migrations are key elements of Atlantic sturgeon life history for the Gulf of Maine DPS as well as likely throughout the entire range (Atlantic Sturgeon Status Review Team 2007; Fernandes et al. 2010).

Historically, the Gulf of Maine DPS likely supported more than 10,000 spawning adults (Atlantic Sturgeon Status Review Team 2007; Maine State Planning Office 1993; Secor 2002). Other than the Northeast Area Monitoring and Assessment Program based estimates presented above, there are no empirical abundance estimates for the Gulf of Maine DPS. The Atlantic Sturgeon Status Review Team (2007) presumed that the Gulf of Maine DPS was comprised of less than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over 2 time periods, 1977-1981 and 1998-2000, resulted in the capture of 9 adult Atlantic sturgeon (Squiers 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the gear used may not have been selective for larger, adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies.

The 2017 Assessment determined the Gulf of Maine DPS abundance is "depleted" relative to historical levels. It also determined there is a 51% probability Gulf of Maine DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a 74% probability the Gulf of Maine DPS is still subjected to mortality levels higher than determined acceptable in the 2017 Assessment (Atlantic States Marine Fisheries Commission 2017).

The 2017 Assessment reported an N_e for the St. Lawrence, St. John, and Kennebec rivers in the Gulf of Maine DPS. In the St. Lawrence, samples from 30 individuals collected in 2013 produced an estimated N_e of 39.0 individuals (Atlantic States Marine Fisheries Commission 2017). In the St. John River, 31 samples were collected from 1991-1993 and produced an estimated N_e of 115.0 individuals (Atlantic States Marine Fisheries Commission 2017). For the Kennebec River, samples from 52 individuals were collected from 1980-2011, and produced an estimated N_e of 63.4 individuals (Atlantic States Marine Fisheries Commission 2017). While not inclusive of all the spawning rivers in the Gulf of Maine DPS, the effective population size estimate for the St. John River suggests that spawning subpopulation may be large enough to avoid inbreeding depression ($N_e < 100$); however, the estimates for the remaining 2 rivers suggests those spawning subpopulations may be at risk. All 3 spawning subpopulations are likely at risk losing evolutionary potential ($N_e < 1000$). The NMFS Greater Atlantic Region NEAMAP model estimates a minimum ocean population for the entire DPS of 7,455 Atlantic sturgeon, of which 1,864 are adults. We estimate the minimum number of individuals from the Gulf of Maine DPS occurring in the Southeast is 9. Given the significant distance between the rivers of this DPS and Southeast, we anticipate these individuals would be adults.

Viability of Atlantic Sturgeon DPSs

The concept of a viable population able to adapt to changing environmental conditions is critical to Atlantic sturgeon, and the low population numbers of every river population in the 5 DPSs on the East Coast put them in danger of extinction throughout their range. None of the riverine spawning subpopulations are large or stable enough to provide with any level of certainty for continued existence of any of the DPSs. Although the largest impact that caused the precipitous decline of the species has been prohibited (directed fishing), the Atlantic sturgeon population sizes within each DPS have remained relatively constant at greatly reduced levels for 100 years. The largest Atlantic sturgeon population in the United States, the Hudson River population within the New York Bight DPS, is estimated to have only 870 spawning adults each year. The Altamaha River population within the SA DPS is the largest Atlantic sturgeon population in the Southeast and only has an estimated 343 adults spawning annually. All other Atlantic sturgeon river populations in the U.S. are estimated to have less than 300 spawning adults annually.

Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry 1971; Shaffer 1981; Soulé 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life span allows multiple opportunities to contribute to future generations, it also increases the time frame over which exposure to the multitude of threats facing Atlantic sturgeon can occur.

The viability of the Atlantic sturgeon DPSs depends on having multiple self-sustaining riverine spawning subpopulations and maintaining suitable habitat to support the various life functions (spawning, feeding, growth) of Atlantic sturgeon populations. Because a DPS is a group of populations, the stability, viability, and persistence of individual populations affects the persistence and viability of the larger DPS. The loss of any population within a DPS will result in: (1) a long-term gap in the range of the DPS that is unlikely to be recolonized; (2) loss of reproducing individuals; (3) loss of genetic biodiversity; (4) potential loss of unique haplotypes; (5) potential loss of adaptive traits; (6) reduction in total number; and (7) potential for loss of population source of recruits. The loss of a population will negatively impact the persistence and viability of the DPS as a whole, as fewer than 2 individuals per generation spawn outside their natal rivers (King et al. 2001; Waldman et al. 2002a; Wirgin et al. 2000). The persistence of individual populations, and in turn the DPS, depends on successful spawning and rearing within the freshwater habitat, the immigration into marine habitats to grow, and then the return of adults to natal rivers to spawn.

4.1.2.4 Threats (All DPSs)

Atlantic sturgeon were once numerous along the East Coast until fisheries for their meat and caviar reduced the populations by over 90% in the late 1800s. Fishing for Atlantic sturgeon became illegal in state waters in 1998 and in remaining U.S. waters in 1999. Dams, dredging, poor water quality, and accidental catch (bycatch) by fishermen continue to threaten Atlantic sturgeon. Though Atlantic sturgeon populations appear to be increasing in some rivers, other river populations along the East Coast continue to struggle and some have been eliminated entirely. The 5 DPSs of Atlantic sturgeon were listed as threatened or endangered under the ESA primarily as a result of a combination of habitat restriction and modification, overutilization (i.e., being taken as bycatch) in commercial fisheries, and the inadequacy of regulatory mechanisms in ameliorating these impacts and threats.

Dams

Dams for hydropower generation, flood control, and navigation adversely affect Atlantic sturgeon by impeding access to spawning, developmental, and foraging habitat, modifying free-flowing rivers to reservoirs, physically damaging fish on upstream and downstream migrations, and altering water quality in the remaining downstream portions of spawning and nursery habitat (Atlantic Sturgeon Status Review Team 2007). Attempts to minimize the impacts of dams using measures such as fish passage have not proven beneficial to Atlantic sturgeon, as they do not regularly use existing fish passage devices, which are generally designed to pass pelagic fish (i.e., those living in the water column) rather than bottom-dwelling species, like sturgeon. However, NMFS continues to evaluate ways to effectively pass sturgeon above and below man-made barriers. For example, large nature-like fishways (e.g., rock ramps) hold promise as a mechanism for successful passage.

Within the range of the Carolina DPS, dams have restricted Atlantic sturgeon spawning and juvenile developmental habitat by blocking over 60% of the historical sturgeon habitat upstream of the dams in the Cape Fear and Santee-Cooper River systems. Water quality (velocity, temperature, and DO downstream of these dams, as well as on the Roanoke River, has been reduced, which modifies and restricts the extent of spawning and nursery habitat for the Carolina DPS.

Within in the range of the SA DPS, on the Savannah River, the New Savannah Bluff Lock and Dam at the City of Augusta, is located just a few kilometers below impassible rapids, denying Atlantic sturgeon access to 7% of its historically available habitat (Atlantic Sturgeon Status Review Team 1998). However, the Augusta Shoals, the only rocky shoal habitat on the Savannah River and the former primary spawning habitat for Atlantic sturgeon in the river (Duncan et al. 2003; Marcy Jr. et al. 2005; USFWS 2003; Wrona et al. 2007), is located above New Savannah Bluff Lock and Dam, and is currently inaccessible to Atlantic sturgeon. So, while Atlantic sturgeon have access to the majority of historical habitat in terms of unimpeded river miles, only a small amount of spawning habitat exists downstream of the New Savannah Bluff Lock and Dam and the vast majority of the rocky freshwater spawning habitat they need is inaccessible as a result of the New Savannah Bluff Lock and Dam.

Within the range of the New York Bight DPS, the Holyoke Dam on the Connecticut River blocks further upstream passage; however, the extent that Atlantic sturgeon historically would have used habitat upstream of Holyoke is unknown. Connectivity may be disrupted by the presence of dams on several smaller rivers in the New York Bight region. Connectivity is disrupted by the presence of dams on several rivers in the range of the Gulf of Maine DPS. Within the Gulf of Maine DPS, access to historical spawning habitat is most severely impacted in the Merrimack River (Atlantic Sturgeon Status Review Team 2007). Construction of the Essex Dam blocked the migration of Atlantic sturgeon to 58% of its historically available habitat (Atlantic Sturgeon Status Review Team 2007). The extent that Atlantic sturgeon are affected by operations of dams in the Gulf of Maine region is currently unknown.

Dredging

Riverine, nearshore, and offshore areas are often dredged to support commercial shipping and recreational boating, construction of infrastructure, and marine mining. Environmental impacts of dredging include the direct removal/burial of prey species; turbidity/siltation effects; contaminant resuspension; noise/disturbance; alterations to hydrodynamic regime and physical habitat; and actual loss of riparian habitat (Chytalo 1996; Winger et al. 2000). According to Smith and Clugston (1997), dredging and filling impact important habitat features of Atlantic sturgeon as they disturb benthic fauna, eliminate deep holes, and alter rock substrates.

In the SA DPS, maintenance dredging is currently modifying Atlantic sturgeon nursery habitat in the Savannah River. Modeling indicates that the proposed deepening of the navigation channel will result in reduced DO and upriver movement of the salt wedge, restricting spawning habitat. Dredging is also modifying nursery and foraging habitat in the St. Johns River. For the Carolina DPS, dredging in spawning and nursery grounds modifies the quality of the habitat and is further restricting the extent of available habitat in the Cape Fear and Cooper Rivers, where Atlantic sturgeon habitat has already been modified and restricted by the presence of dams. Dredging for navigational purposes is suspected of having reduced available spawning habitat for the Chesapeake Bay DPS in the James River (Atlantic Sturgeon Status Review Team 2007; Bushnoe et al. 2005; Holton Jr. and Walsh 1995). Both the Hudson and Delaware rivers have navigation channels that are maintained by dredging. Dredging is also used to maintain channels in the nearshore marine environment. Many rivers in the range of the Gulf of Maine DPS, including the Kennebec River, also have navigation channels that are maintained by dredging. Dredging outside of federal channels and in-water construction occurs throughout the range of the Chesapeake Bay, New York Bight and Gulf of Maine DPSs.

Dickerson (2013) summarized observed takings of 38 sturgeon from dredging activities conducted by USACE and observed from 1990-2013: 3 Gulf, 11 shortnose, and 23 Atlantic, along the Atlantic Coast (NMFS Greater Atlantic Region and NMFS Southeast Region). Five of those shortnose sturgeon were reported entrained at the disposal sites in the NMFS Greater Atlantic Region from 1996-1998 (NMFS 2018d). All 5 takes occurred while dredging in known overwintering aggregation areas, likely increasing the risk of entrainment.

Of the 3 types of dredges considered by Dickerson (2013) (hopper, clamshell, and pipeline), most sturgeon were captured by hopper dredge, though some takes were also noted in clamshell and pipeline dredges. Notably, reports include only those trips when an observer was on board to document capture. Additional data provided by USACE indicate another 16 Atlantic sturgeon were killed by hopper dredging from 2016-2018 in the Southeast. To offset the adverse effects associated dredging relocation trawling is used at times. The USACE has used this technique during dredging at Brunswick Harbor, Savannah Harbor, Kings Bay, and in the Savannah River channel. Trawling in these area captured 215 and relocated 215 Atlantic sturgeon from 2016-2018.

Water Quality

Atlantic sturgeon rely on a variety of water quality parameters to successfully carry out their life functions. Low DO and the presence of contaminants modify the quality of Atlantic sturgeon habitat and in some cases, restrict the extent of suitable habitat for life functions. Secor (1995) noted a correlation between low abundances of sturgeon during this century and decreasing water quality caused by increased nutrient loading and increased spatial and temporal frequency of hypoxic (low oxygen) conditions. Of particular concern is the high occurrence of low DO coupled with high temperatures in the river systems throughout the range of the Carolina and SA DPSs in the Southeast. Sturgeon are more highly sensitive to low DO than other fish species (Niklitschek and Secor 2009c) and low DO in combination with high temperature is particularly problematic for Atlantic sturgeon. Studies have shown that juvenile Atlantic sturgeon experience lethal and sublethal (metabolic, growth, feeding) effects as DO drops and temperatures rise (Niklitschek and Secor 2005; Niklitschek and Secor 2009c; Secor and Gunderson 1998).

Reductions in water quality from terrestrial activities have modified habitat utilized by the SA DPS. Low DO is modifying sturgeon habitat in the Savannah due to dredging, and non-point source inputs are causing low DO in the Ogeechee River and in the St. Marys River, which completely eliminates juvenile nursery habitat in summer. Low DO has also been observed in the St. Johns River in the summer. In the Pamlico and Neuse systems of the Carolina DPS, nutrient-loading and seasonal anoxia are occurring, associated in part with concentrated animal feeding operations. Heavy industrial development and concentrated animal feeding operations have degraded water quality in the Cape Fear River. Water quality in the Waccamaw and Yadkin-Pee Dee Rivers has been affected by industrialization and riverine sediment samples contain high levels of various toxins, including dioxins.

Decreased water quality also threatens Atlantic sturgeon of the Chesapeake Bay DPS, especially since the Chesapeake Bay system is vulnerable to the effects of nutrient enrichment due to a relatively low tidal exchange and flushing rate, large surface-to-volume ratio, and strong stratification during the spring and summer months (Atlantic States Marine Fisheries Commission 1998; Atlantic Sturgeon Status Review Team 2007; Pyzik et al. 2004). These conditions contribute to reductions in DO levels throughout the bay. The availability of nursery habitat, in particular, may be limited given the recurrent hypoxia (low DO) conditions within the Bay (Niklitschek and Secor 2005; Niklitschek and Secor 2010).

Both the Hudson and Delaware Rivers, as well as other rivers in the New York Bight region, were heavily polluted in the past from industrial and sewer discharges. In the past, many rivers in Maine, including the Androscoggin River, were heavily polluted from industrial discharges from pulp and paper mills. While water quality has improved and most discharges are limited through regulations, many pollutants persist in the benthic environment of the New York Bight and Gulf of Maine DPSs. It is particularly problematic if pollutants are present on spawning and nursery grounds, as developing eggs and larvae are particularly susceptible to exposure to contaminants.

Atlantic sturgeon may also be particularly susceptible to impacts from environmental contamination because they are long-lived, benthic feeders. Sturgeon feeding in estuarine habitats near urbanized areas may be exposed to numerous suites of contaminants within the substrate. Contaminants, including toxic metals, polychlorinated aromatic hydrocarbons, organophosphate and organochlorine pesticides, PCBs, and other chlorinated hydrocarbon compounds can have substantial deleterious effects on aquatic life. These elements and compounds can cause acute lesions, growth retardation, and reproductive impairment in fishes (Atlantic Sturgeon Status Review Team 2007; Cooper 1989; Sindermann 1994).

Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but their long-term effects are not known (Ruelle and Henry 1992; Ruelle and Keenlyne 1993). Elevated levels of contaminants, including chlorinated hydrocarbons, in several other fish species are associated with reproductive impairment (Cameron et al. 1992; Drevnick and Sandheinrich 2003; Hammerschmidt et al. 2002; Longwell et al. 1992), reduced egg viability (Billsson et al. 1998; Giesy et al. 1986; Mac and Edsall 1991; Matta et al. 1997; von Westernhagen et al. 1981), reduced survival of larval fish (Berlin et al. 1981; Giesy et al. 1986), delayed maturity (Jorgensen (Jørgensen et al. 2004) and posterior malformations (Billsson et al. 1998). Pesticide exposure in fish may affect antipredator and homing behavior, reproductive function, physiological development, and swimming speed and distance (Beauvais et al. 2000; Moore and Waring 2001; Scholz et al. 2000; Waring and Moore 2004). It should be noted that the effect of multiple contaminants or mixtures of compounds at sublethal levels on fish has not been adequately studied. Atlantic sturgeon use marine, estuarine, and freshwater habitats and are in direct contact through water, diet, or dermal exposure with multiple contaminants throughout their range (Atlantic Sturgeon Status Review Team 2007). Trace metals, trace elements, or inorganic contaminants (mercury, cadmium, selenium, lead, etc.) are another suite of contaminants occurring in fish. Post (1987) states that toxic metals may cause death or sublethal effects to fish in a variety of ways and that chronic toxicity of some metals may lead to the loss of reproductive capabilities, body malformation, inability to avoid predation, and susceptibility to infectious organisms.

Water Quantity

Water allocation issues are a growing threat in the Southeast and exacerbate existing water quality problems. Taking water from one basin and transferring it to another fundamentally and irreversibly alters natural water flows in both the originating and receiving basins, which can affect DO levels, temperature, and the ability of the basin of origin to assimilate pollutants (GWC 2006a). Water quality within the river systems in the range of the South Atlantic and Carolina DPSs is negatively affected by large water withdrawals. Known water withdrawals of over 240 million gallons per day are permitted from the Savannah River for power generation and municipal uses. However, permits for users withdrawing less than 100,000 gallons per day are not required, so actual water withdrawals from the Savannah and other rivers within the range of the SA DPS are likely much higher.

In the range of the Carolina DPS, 20 interbasin water transfers in existence prior to 1993, averaging 66.5 million gallons per day, were authorized at their maximum levels without being subjected to an evaluation for certification by the North Carolina Department of Environment and Natural Resources or other resource agencies. Since the 1993 legislation requiring certificates for transfers, almost 170 million gallons per day of interbasin water withdrawals have been authorized, with an additional 60 million gallons per day, pending certification. The removal of large amounts of water from these systems will alter flows, temperature, and DO. Water shortages and “water wars” are already occurring in the rivers occupied by the SA and Carolina DPSs and will likely be compounded in the future by population growth and potentially by climate change.

Climate Change

Large-scale factors impacting riverine water quality and quantity that likely exacerbate habitat threats to Atlantic sturgeon of all 5 DPSs include drought, and intra- and inter-state water allocation. Changes in the climate are very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. For example, annual precipitation in the Southeast has increased by 0.19 inches per decade since 1950 (NOAA 2018a) but has also experienced several significant (i.e., categorized as “abnormally dry” to “exceptional”) since 2000 (National Drought Mitigation Center 2018). The Northeast has seen even more significant increases in annual precipitation with increases of 0.71 inches per decade since 1950 (NOAA 2018a). While not as severe, the Northeast has also experienced 2 periods of notable drought since 2000, as well as multiple other dry periods during that period. Abnormally low stream flows can restrict access by sturgeon to habitat areas and exacerbate water quality issues such as water temperature, reduced DO, nutrient levels, and contaminants.

Long-term observations also confirm changes in temperature are occurring at a rapid rate. From 1895-2017, the average annual temperature in the Southeast has risen 0.1°F per decade. From 1950-2017, the increase triples to 0.3°F per decade per decade (NOAA 2018a). Increases in the average annual temperature are even greater in the Northeast where even greater. From 1895-2017, the average annual temperature rose 0.2°F per decade. From 1950-2017, the increase was the same as in the Southeast, 0.3°F per decade per decade (NOAA 2018a). Aside from observation, climate modeling also projects future increases in temperatures in both the Southeast and Northeast. Table 33 summarizes the projected increases by the mid-century (2036–2065) and late-century (2071–2100). These are projections from the Representative Concentration Pathway (RCP) model scenarios RCP8.5 and RCP4.5, used by the Intergovernmental Panel on Climate Change), relative to average from 1976–2005 (Hayhoe et al. 2017).[48]

Table 33. Projected Temperature Increase in the Southeast and Northeast Under Two Model Projections and Time Series (Hayhoe et al. 2017)

National Climate Assessment Region	RCP4.5 Mid-Century (2036–2065)	RCP8.5 Mid-Century (2036–2065)	RCP4.5 Late-Century (2071–2100)	RCP8.5 Late-Century (2071–2100)
Northeast	3.98°F (2.21°C)	5.09°F (2.82°C)	5.27°F (2.92°C)	9.11°F (5.06°C)
Southeast	3.40°F (1.89°C)	4.30°F (2.39°C)	4.43°F (2.46°C)	7.72°F (4.29°C)

Ocean temperature in the U.S. Northeast Shelf and surrounding Northwest Atlantic waters has increased faster than the global average over the last decade (Pershing et al. 2015). New projections for the U.S. Northeast Shelf and Northwest Atlantic Ocean suggest that this region will warm 2 to 3 times faster than the global average (Saba et al. 2016). A first-of-its-kind climate vulnerability assessment, conducted on 82 fish and invertebrate species in the Northeast U.S. Shelf, concluded that Atlantic sturgeon from all 5 DPSs were among the most vulnerable species to global climate change (Hare et al. 2016).

Sea-level rise is another consequence of climate change; it has already had significant impacts on coastal areas and these impacts are likely to increase. Since 1852, when the first topographic maps of the Southeastern United States were prepared, high tidal flood elevations have increased approximately 12 inches. During the 20th century, global sea level has increased 15 to 20 cm (National Assessment Synthesis Team 2000). Sea level rise is also projected to extend areas of salinization of groundwater and estuaries. Some of the most populated areas of this region are low-lying; the threat of saltwater entering into this region's aquifers with projected sea level rise is a concern (U.S. Global Change Research Program 2001). Saltwater intrusion will likely exacerbate existing water allocation issues, leading to an increase in reliance on interbasin water transfers to meet municipal water needs, further stressing water quality. Similarly, saltwater intrusion is likely to affect local ecosystems. Analysts attribute the forest decline in the Southeast to saltwater intrusion associated with sea level rise. Coastal forest losses will be even more severe if sea level rise accelerates as is expected as a result of global warming.

[48] RCPs make predictions based on changes, if any, in future greenhouse gas emissions. Specifically, they evaluate radiative forcing, or the amount of energy stored at the Earth's surface in watts/m^2. As the amount of greenhouse gases increases in the atmosphere more energy is trapped, and the number of watts/m^2 increases. RCP2.6 and RCP8.5 represent the lowest and highest radiative scenarios, of 2.6 watts/m^2 and 8.5 watts/m^2, respectively. RCP4.5 and RCP6.0 assume intermediate levels of radiative forcing.

The effects of future climate change will vary greatly in diverse coastal regions for the United States. Warming is very likely to continue in the United States during the next 25 to 50 years, regardless of reduction in greenhouse gases, due to emissions that have already occurred (National Assessment Synthesis Team 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25 to 50 years, and it is possible that they will accelerate. A warmer and drier climate would reduce stream flows and increase water temperatures. Expected consequences would be a decrease in the amount of DO in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer, wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch et al. 2000).

Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the Southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer et al. 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change.

In marine waters, there is a high confidence that observed changes are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (Intergovernmental Panel on Climate Change 2007).

Although Atlantic sturgeon have persisted for millions of years and have experienced wide variations in global climate conditions, the current rate of climate change reported and/or anticipated to occur is faster than what we can reasonably expect Atlantic sturgeon to be able to adapt to.

Vessel Strikes

Vessel strikes are a threat to the Chesapeake Bay and New York Bight DPSs. Eleven Atlantic sturgeon were reported to have been struck by vessels on the James River from 2005 through 2007. Several of these were mature individuals. From 2007–2010, researchers documented 31 carcasses of adult Atlantic sturgeon in the tidal freshwater portion of the James River, Virginia (Balazik et al. 2012b). Twenty-six of the carcasses had gashes from vessel propellers, and the remaining 5 carcasses were too decomposed to allow determination of the cause of death (Balazik et al. 2012b). The types of vessels responsible for these mortalities could not be confirmed. Most (84%) of the carcasses were found in a relatively narrow reach that has been modified to increase shipping efficiency (Balazik et al. 2012b). Using telemetry, Balazik et al. (2012b) reported that while staging (holding in an area from hours to days, with minimal upstream or downstream movements), adult male Atlantic sturgeon spent most (62%) of their time within 1 m of the river bottom. Under the assumption that Atlantic sturgeon do not modify their behavior as a result of vessel noise, Balazik et al. (2012b) hypothesized adult male Atlantic sturgeon in the James River would rarely encounter small recreational boats or tugboats with shallow drafts. Instead, Balazik et al. (2012b) concluded vessel strike mortalities are likely caused by deep-draft ocean cargo ships, with drafts that coincide with the river depths most frequently used by the animals they tracked using telemetry. Ultimately, they estimated that current monitoring in the James River documents less than one-third of vessel strike mortalities (Balazik et al. 2012b).

From 2004-2008, 29 mortalities believed to be the result of vessel strikes were documented in the Delaware River; at least 13 of these fish were large adults. The time of year when these events occurred (predominantly May through July, with 2 in August), indicate the animals were likely adults migrating through the river to the spawning grounds. Because we do not know the percent of total vessel strikes that these observed mortalities represent, we are not able to quantify the number of individuals likely killed as a result of vessel strikes in the Chesapeake and New York Bight DPSs. Very little is known about the effects of vessel strikes on individuals from the Carolina or SA DPSs.

Bycatch Mortality

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to Atlantic sturgeon in all 5 DPSs. Atlantic sturgeon are more sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum reproductive rates, and a large percentage of egg production occurs later in life. Based on these life history traits, Boreman (1997) calculated that Atlantic sturgeon can only withstand the annual loss of up to 5% of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0% and 51%, with the greatest mortality occurring in sturgeon caught by sink gillnets. Currently, there are estimates of the number of Atlantic sturgeon captured and killed in sink gillnet and otter trawl fisheries authorized by Fishery Management Plans in the Northeast Region (Miller and Shepherd 2011). Those estimates indicate from 2006-2010, on average there were 1,548 and 1,569 encounters per year in observed gillnet and trawl fisheries, respectively, with an average of 3,118 encounters combined annually. Mortality rates in gillnet gear were approximately 20%, while mortality rates in otter trawl gear are generally lower, at approximately 5%. Atlantic sturgeon are particularly vulnerable to being caught in sink gillnets; therefore, fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch. Atlantic sturgeon are incidentally captured in state and federal fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Atlantic States Marine Fisheries Commission 2007; Stein et al. 2004a). Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

Stochastic Events

Stochastic events, such as hurricanes, are common throughout the range of Atlantic sturgeon from all 5 DPSs. These events are unpredictable and their effect on the survival and recovery of the species in unknown; however, they have the potential to impede the survival and recovery directly if animals die as a result of them, or indirectly if habitat, is damaged as a result of these disturbances. For example, in 2018, flooding from Hurricane Florence flushed significant amounts of organic matter into rivers supporting sturgeon in the Southeast. The DO levels in those rivers dropped so low (i.e., 0.2 mg/L) that thousands of fish suffocated, including multiple sturgeon.

4.1.3 Shortnose sturgeon

Shortnose sturgeon were initially listed as an endangered species by USFWS on March 11, 1967, under the Endangered Species Preservation Act (32 FR 4001). Shortnose sturgeon continued to meet the listing criteria as "endangered" under subsequent definitions specified in the 1969 Endangered Species Conservation Act and remained on the list with the inauguration of the ESA in 1973. NMFS assumed jurisdiction for shortnose sturgeon from USFWS in 1974 (39 FR 41370). The shortnose sturgeon currently remains listed as an endangered species throughout all of its range along the east coast of the United States and Canada. A recovery plan for shortnose sturgeon was published by NMFS in 1998 (NMFS 1998).

4.1.3.1 Species Description and Distribution

The shortnose sturgeon (*Acipenser brevirostrum)* is the smallest of the 3 sturgeon species that occur in eastern North America. They attain a maximum length of about 6 feet, and a weight of about 55 pounds. Shortnose sturgeon inhabit large coastal rivers of eastern North America. Although considered an amphidromous species,[49] shortnose sturgeon are more properly characterized as "freshwater amphidromous," meaning that they move between fresh and salt water during some part of their life cycle, but not necessarily for spawning. Shortnose sturgeon rarely leave the rivers where they were born ("natal rivers"). Shortnose sturgeon feed opportunistically on benthic insects, crustaceans, mollusks, and polychaetes (Dadswell et al. 1984).

[49] Meaning they are born in freshwater, then live primarily in their natal river, making short feeding or migratory trips into salt water, and then return to freshwater.

Historically, shortnose sturgeon were found in the coastal rivers along the east coast of North America from the Saint John River, New Brunswick, Canada, to the St. Johns River, Florida, and perhaps as far south as the Indian River in Florida (Evermann and Bean 1898; Gilbert 1989b). Currently, the distribution of shortnose sturgeon across their range is disconnected, with northern populations separated from southern populations by a distance of about 250 miles (400 km) near their geographic center in Virginia (see Figure 3.2). In the southern portion of the range, they are currently found in the Edisto, Cooper, Altamaha, Ogeechee, and Savannah Rivers in Georgia. Sampling has also found shortnose in the Roanoke River, Albemarle Sound, and Cape Fear Rivers, while fishers have reported the species in Neuse River and Pamlico Sound (NMFS 2010a). Females bearing eggs have been collected in the Cape Fear River (Moser and Ross 1995). Spawning is known to be occurring in the Cooper River (NMFS 2010a; Ruddle 2018)), the Congaree River (Collins et al. 2003; Post et al. 2017b), and the Pee Dee River (NMFS 2010a). While it had been concluded that shortnose sturgeon are extinct from the Satilla River in Georgia and the St. Marys River along the Florida and Georgia border, targeted surveys in both the Satilla (Fritts and Peterson 2010) and St. Marys (Fox and Peterson 2017; Fritts and Peterson 2010) have captured shortnose sturgeon. A single specimen was found in the St. Johns River by the Florida Fish and Wildlife Conservation Commission during extensive sampling of the river in 2002 and 2003 (NMFS 2010a).

4.1.3.2 Life History Information

Shortnose sturgeon populations show clinal variation, [50] with a general trend of faster growth and earlier age at maturity in more southern systems. Fish in the southern portion of the range grow the fastest, but growth appears to plateau over time. Conversely, fish in the northern part of the range tend to grow more slowly, but reach a larger size because they continue to grow throughout their lives. Male shortnose sturgeon mature at 2-3 years of age in Georgia, 3-5 years of age in South Carolina, and 10-11 years of age in the Saint John River, Canada. Females mature at 4-5 years of age in Georgia, 7-10 years of age in the Hudson River, New York, and 12-18 years of age in the Saint John River, Canada. Because animals are considered mature at the onset of developing mature gonads, spawning is usually delayed relative to reaching maturity. Males begin to spawn 1-2 years after reaching sexual maturity and spawn every 1-2 years (Dadswell 1979; Kieffer and Kynard 1996; NMFS 1998). Age at first spawning for females is about 5 years post-maturation with spawning occurring every 3-5 years (Dadswell 1979). Fecundity of shortnose sturgeon ranges between approximately 30,000-200,000 eggs per female (Gilbert 1989b).

Adult shortnose sturgeon spawn in the rivers where they were born. Initiation of the upstream movement of shortnose sturgeon to spawn is likely triggered partially by water temperatures. Shortnose sturgeon captured in 5 coastal river systems of South Carolina all spawned during temperatures ranges from 5–18°C (Post et al. 2014a), which is similar to what has been documented throughout the range (Duncan et al. 2004; Hall et al. 1991; Kieffer and Kynard 1996; NMFS 1998; Taubert 1980). In the Altamaha River, Georgia, adults began their upstream migrations during likely spawning runs during the late-winter months when water temperatures declined to 11.6–16.9 °C (Post et al. 2014a). Water depth and flow are also important at

[50] A gradual change in a character or feature across the distributional range of a species or population, usually correlated with an environmental or geographic transition

spawning sites (Kieffer and Kynard 1996). Spawning sites are characterized by moderate river flows with average bottom velocities between 1-2.5 ft (0.4-0.8 m) per second (Hall et al. 1991; Kieffer and Kynard 1996; NMFS 1998). Shortnose sturgeon tend to spawn on rubble, cobble, or large rocks (Buckley and Kynard 1985; Dadswell 1979; Kynard 1997; Taubert 1980), timber, scoured clay, or gravel (Hall et al. 1991). Southern populations of shortnose sturgeon usually spawn at least 125 miles (200 km) upriver (Kynard 1997) or throughout the fall line[51] zone if they are able to reach it. Adults typically spawn in the late winter to early spring (December-March) in southern rivers (i.e., North Carolina and south) and the mid to late spring in northern rivers. They spend the rest of the year in the vicinity of the saltwater/freshwater interface (Collins and Smith 1993).

Little is known about young-of-the-year behavior and movements in the wild, but shortnose sturgeon at this age are believed to remain in channel areas within freshwater habitats upstream of the saltwater/freshwater interface for about 1 year, potentially due to their low tolerance for salinity (Dadswell et al. 1984; Kynard 1997). Residence of YOY in freshwater is supported by several studies on cultured shortnose sturgeon (Jarvis et al. 2001; Jenkins et al. 1993; Ziegeweid et al. 2008). In most rivers, juveniles aged 1 and older join adults and show similar patterns of habitat use (Kynard 1997). In the Southeast, juveniles aged 1 year and older make seasonal migrations like adults, moving upriver during warmer months where they shelter in deep holes, before returning to the fresh/saltwater interface when temperatures cool (Collins et al. 2002; Flournoy et al. 1992a). Due to their low tolerance for high temperatures, warm summer temperatures (above 82°F) may severely limit available juvenile rearing habitat in some rivers in the southeastern United States. Juveniles in the Saint John, Hudson, and Savannah Rivers use deep channels over sand and mud substrate for foraging and resting (Dovel et al. 1992; Hall et al. 1991; Pottle and Dadswell 1979).

4.1.3.3 Status and Population Dynamics

The 1998 shortnose sturgeon recovery plan identified 19 distinct shortnose sturgeon populations based on natal rivers (NMFS 1998). Since 1998, significantly more tagging/tracking data on straying rates to adjacent rivers has been collected, and several genetic studies have determined where coastal migrations and effective movement (i.e., movement with spawning) are occurring. Genetic analyses aided in identifying population structure across the range of shortnose sturgeon. Several studies indicate that most, if not all, shortnose sturgeon riverine populations are statistically different ($p < 0.05$) (King et al. 2001; Waldman et al. 2002b; Wirgin et al. 2005; Wirgin et al. 2010; Wirgin et al. 2000). Gene flow is low between riverine populations indicating that while shortnose sturgeon tagged in one river may later be recaptured in another, it is unlikely the individuals are spawning in those non-natal rivers. This is consistent with our knowledge that adult shortnose sturgeon are known to return to their natal rivers to spawn (NMFS 1998). However, Fritts et al. (2016) provide evidence that greater mixing of riverine populations occurs in areas where the distance between adjacent river mouths is relatively close, such as in the Southeast.

[51] The fall line is the boundary between an upland region of continental bedrock and an alluvial coastal plain, sometimes characterized by waterfalls or rapids.

Aside from genetic differences associated with shortnose sturgeon only spawning in their natal rivers, researchers have also identified levels of genetic differentiation that indicate high degrees of reproductive isolation in at least 3 groupings (i.e., metapopulations) (Figure 30). Shortnose sturgeon in the Southeast comprise a single metapopulation, the “Carolinian Province” (Figure 30) Wirgin et al. (2010) note that genetic differentiation among populations within the Carolinian Province was considerably less pronounced than among those in the other 2 metapopulations (i.e., Virginian Province and Acadian Province) and contemporary genetic data suggest that reproductive isolation among these populations is less than elsewhere. In other words, the shortnose sturgeon populations within the Carolinian Province are more closely related to each other, than the populations that make up either the Virginian or Acadian Provinces.

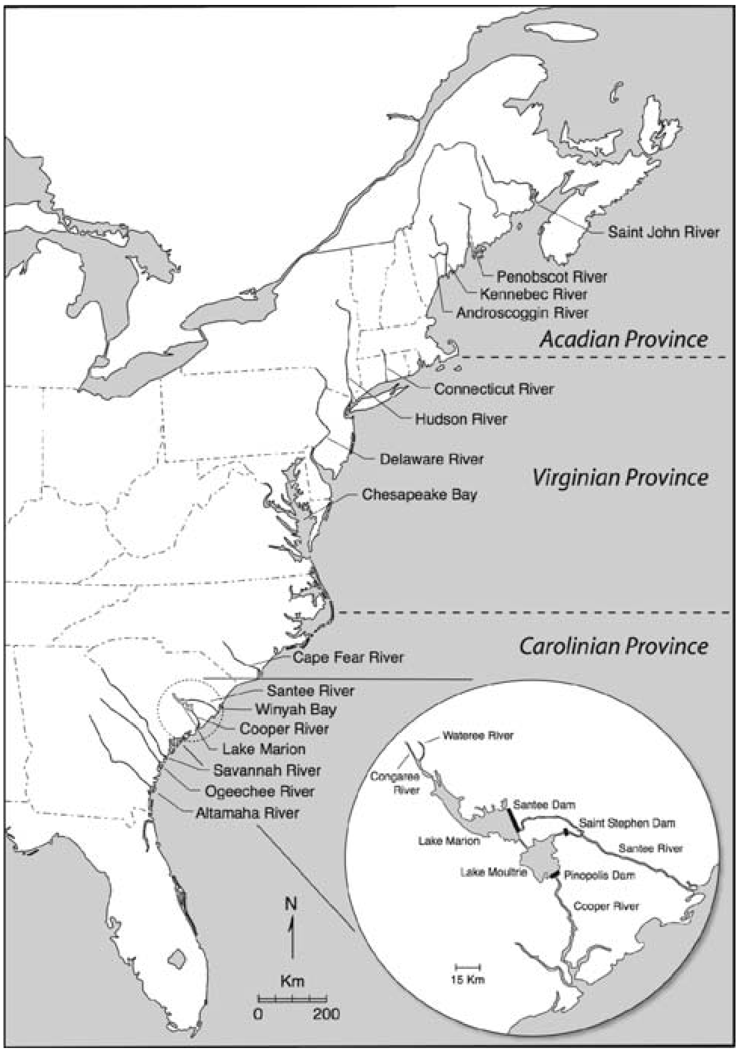

Figure 30. The North American Atlantic coast depicting 3 shortnose sturgeon metapopulations based on mitochondrial DNA control region sequence analysis (Wirgin et al. 2010).

The 3 shortnose sturgeon metapopulations should not be considered collectively but as individual units of management because each is reproductively isolated from the other and constitutes an evolutionarily (and likely an adaptively) significant lineage. Loss of the Carolinian Province ("southern") metapopulation of shortnose sturgeon would result in the loss of the southern half of the species' range (i.e., there is no known reproduction occurring between the Delaware River and Winyah Bay, SC). Loss of the Virginian Province ("mid-Atlantic") metapopulation would create a conspicuous discontinuity in the range of the species from the Hudson River to the northern extent of the southern metapopulation. The Acadian Province ("northern") metapopulation constitutes the northernmost portion of the U.S. range. Loss of the mid-Atlantic metapopulation (Virginian Province) would create a conspicuous discontinuity in the range of the species from the Hudson River to the northern extent of the southern metapopulation. The northern metapopulation constitutes the northernmost portion of the U.S. range. Loss of this metapopulation would result in a significant gap in the range that would serve to isolate the shortnose sturgeon that reside in Canada from the remainder of the species' range in the United States. The loss of any metapopulation would result in a decrease in spatial range, biodiversity, unique haplotypes, adaptations to climate change, and gene plasticity. Loss of unique haplotypes that may carry geographic specific adaptations would lead to a loss of genetic plasticity and, in turn, decrease adaptability. The loss of any metapopulation would increase species' vulnerability to random events.

The current status of the shortnose sturgeon in the Southeast is variable. Populations within the southern metapopulation are relatively small compared to their northern counterparts. Table 34 shows available abundance estimates for rivers in the Southeast. The Altamaha River supports the largest known shortnose sturgeon population in the Southeast with successful self-sustaining recruitment. Total population estimates in the Altamaha show large interannual variation is occurring; estimates have ranged from as low as 468 fish in 1993 to over 5,550 fish in 2006 (NMFS 1998; Peterson and Bednarski 2013). Abundance estimates for the Ogeechee River indicate the shortnose sturgeon population in this river is considerably smaller than in the Altamaha River. The highest point estimate since 1993 occurred in 2007 and resulted in a total Ogeechee River population estimate of 404 shortnose sturgeon (95% confidence interval [CI]: 175-633) (Peterson and Farrae 2011). However, subsequent sampling in 2008 and 2009 resulted in point estimates of 264 (95% CI: 126-402) and 203 (95% CI: 32-446), respectively (Peterson and Farrae 2011). Spawning is also occurring in the Savannah, Cooper, Congaree, and Yadkin-Pee Dee Rivers. The Savannah River shortnose sturgeon population is possibly the second largest in the Southeast with highest point estimate of the total population occurring in 2013 at 2,432 (95% CI: 1,025-6,102). Mean population estimates were lower in 2014 and 2015, reaching an estimated 1,390 (95% CI: 890-2,257) total individuals in 2015 (Bahr and Peterson 2017). Animals in the Savannah River face many environmental stressors and spawning is likely occurring in only a very small area. While active spawning is occurring in South Carolina's Winyah Bay complex (Black, Sampit, Yadkin-Pee Dee, and Waccamaw Rivers) the population status there is unknown. The most recent estimate for the Cooper Rivers suggests a population of approximately 220 spawning adults (Cooke et al. 2004). Status of the other riverine populations supporting the southern metapopulation is unknown due to limited survey effort, with capture in some rivers limited to less than 5 specimens.

Table 34. Shortnose Sturgeon Populations and Their Estimated Abundances

Population (Location)	**Data Series**	**Abundance Estimate (CI)a**	**Population Segment**	**Reference**
Cape Fear River (NC)		>50	Total	
Winyah Bay (NC, SC)		unknown		

Population (Location)	Data Series	Abundance Estimate (CI)a	Population Segment	Reference
Santee River (SC)		unknown		
Cooper River (SC)	1996-1998	220 (87-301)	Adults	Cooke et al. (2004)
ACE Basin (Ashepoo, Combahee, and Edisto Rivers) (SC)		unknown		
Savannah River (SC, GA)		1,000 - 3,000	Adults	B. Post, SCDNR 2003; NMFS unpublished
	2013	2,432 (1,025-6,102)	Total	Bahr and Peterson (2017)
	2014	1,957 (1,261-3,133)	Total	
	2015	1,390 (890-2,257)	Total	
Ogeechee River (GA)	1993	361 (326-400)	Total	Rogers and Weber (1994);
	1999-2000	147 (104-249)	Total	Fleming et al. (2003)
	2007	404 (175-633)	Total	Peterson and Farrae (2011)
	2008	264 (126-402)	Total	
	2009	203 (32-446)	Total	
Altamaha River (GA)	1988	2,862 (1,069-4,226)	Total	NMFS (1998)
	1990	798 (645-1,045)	Total	NMFS (1998)
	1993	468 (316-903)	Total	NMFS (1998)
	2006	5,551 (2,804–11,304)	Total	(Peterson and Bednarski 2013)
	2009	1,206 (566–2,759)	Total	(Peterson and Bednarski 2013)
Satilla River (GA)		N/A		
Saint Marys River (FL)		N/A		
St. Johns River (FL)		unknown	Total	Fox et al. (2017)

Annual variation in population estimates in many basins is due to changes in yearly capture rates that are strongly correlated with weather conditions (e.g., river flow, water temperatures). In "dry years," fish move into deep holes upriver of the saltwater/freshwater interface, which can make them more susceptible to gillnet sampling. Consequently, rivers with limited data sets among years and limited sampling periods within a year may not offer a realistic representation of the size or trend of the shortnose sturgeon population in the basin. As a whole, the data on shortnose sturgeon populations is rather limited and some of the differences observed between years may be an artifact of the models and assumptions used by the various studies.

4.1.3.4 Threats

The shortnose sturgeon was listed as endangered under the ESA as a result of a combination of habitat degradation or loss (resulting from dams, bridge construction, channel dredging, and pollutant discharges), mortality (from impingement on cooling water intake screens, turbines, climate change, dredging, and incidental capture in other fisheries), and the inadequacy of regulatory mechanisms in ameliorating these impacts and threats. The primary threats to the species today are described below.

Dams

Dams for hydropower generation, flood control, and navigation adversely affect shortnose sturgeon habitat by impeding access to spawning, developmental, and foraging habitat, modifying free-flowing rivers to reservoirs, physically damaging fish on upstream and downstream migrations, and altering water quality in the remaining downstream portions of spawning and nursery habitat.

Historically, sturgeon ascended to the farthest freshwater reaches and river heads to reach natal spawning grounds (Hightower 1998; Lawson 1709; McDonald 1887). An inability to move above dams and use potentially beneficial habitats may restrict population growth (NMFS 1998). Dams blocking migration could force sturgeon to spawn at locations that were not historically used (Kynard et al. 1999). If sturgeon have to deposit eggs in habitat further downstream because of an upstream dam, this may make survival of their progeny less likely. Sturgeon embryos and larvae have limited salt tolerance, so their habitat must be well upstream of the salt front (van Eenennaam et al. 1996). Also, if sturgeon must utilize habitat that is not suitable or less suitable than the original blocked spawning sites for successful adherence, fertilization, and development, then those eggs may not become viable progeny. This will affect the survival and recruitment of individuals of that particular year class and, over time, reduce the reproductive success and recruitment of new individuals to the population.

Fish passage devices have shown limited benefit to shortnose sturgeon as a means of minimizing impacts of dams because these devices have been historically designed for salmon and other water-column fish rather than large, bottom-dwelling species like sturgeon. However, NMFS continues to evaluate ways to effectively pass sturgeon above and below man-made barriers. For example, large nature-like fishways (e.g., rock ramps) hold promise as a mechanism for successful passage. Dams have separated the shortnose sturgeon population in the Cooper River, trapping some above the structure while blocking access upstream to sturgeon below the dam. Telemetry studies indicate that some shortnose sturgeon do pass upriver through the vessel lock in the Pinopolis Dam on the Cooper River in the Santee Cooper Lakes (Post et al. 2014b). In 2011, 2 tagged shortnose sturgeon used the vessel lock in the Pinopolis Dam to pass upstream of the dam. One of the sturgeon was still inhabiting the lakes as of 2013, while the other sturgeon entered Lake Moultrie in March and returned to the Cooper River in April, either through the Pinopolis Lock or through the turbines at Jefferies Power Station (Post et al. 2014b). Shortnose sturgeon inhabit only Lake Marion, the upper of the 2 reservoirs. There is currently no estimate for the portion of the population that inhabits the reservoirs and rivers above the dam.

Additional impacts from dams include the Kirkpatrick Dam (aka Rodman Dam) located about ~12.9 km upstream from the St. Johns River, Florida on the Ocklawaha River (the largest tributary) as part of the Cross Florida Barge Canal. The Ocklawaha River has been speculated as the spawning area for shortnose sturgeon (NMFS 2010a). The New Savannah Bluff Lock and Dam located on the Savannah River on the South Carolina and Georgia border also impedes shortnose sturgeon from accessing upstream shoal areas (NMFS 2010a).

The presence of the dams on the Savannah River also harms sturgeon by restricting life functions other than spawning, particularly in the case of shortnose sturgeon. Sturgeon migrate to optimize feeding, avoid unfavorable conditions, and to optimize reproductive success (McKeown 1984;

Northcote 1978; Tsyplakov 1978). Shortnose sturgeon are considered freshwater amphidromous species and are relatively constrained in their migratory patterns, as they typically migrate between freshwater and mesohaline river reaches but do not migrate extensively to marine habitats for feeding (Kynard 1997).

Dredging

Riverine, nearshore, and offshore areas are often dredged to support commercial shipping and recreational boating, construction of infrastructure, and marine mining. Environmental impacts of dredging include the direct removal/burial of prey species; turbidity/siltation effects; contaminant resuspension; noise/disturbance; alterations to hydrodynamic regime and physical habitat; and actual loss of riparian habitat (Chytalo 1996; Winger et al. 2000). Dredging in spawning and nursery grounds modifies the quality of the habitat and further restricts the extent of available habitat in the Cooper and Savannah Rivers, where shortnose sturgeon habitat has already been modified and restricted by the presence of dams.

Dredging directly effects sturgeon by entraining them in dredge drag arms and impeller pumps. Mechanical dredges have also been documented to kill sturgeon. Dickerson (2013) summarized observed takings of 38 sturgeon from dredging activities conducted by USACE and observed from 1990-2013: 3 Gulf, 11 shortnose, and 23 Atlantic, and 1 unidentified due to decomposition. Of the 3 types of dredges included (hopper, clamshell, and pipeline) in the report, most sturgeon were captured by hopper dredge, though some takes were also noted in clamshell and pipeline dredges. Notably, reports include only those trips when an observer was on board to document capture. To offset the adverse effects associated dredging relocation trawling is used at times. The USACE has successfully used this technique to relocated Atlantic sturgeon, but only 2 shortnose sturgeon (1992 and 2004) have been captured in the Southeast.

Seasonal restrictions on dredging operations have been imposed in some rivers for some species; for example, a March 16–May 31 prohibition to protect striped bass in the Savannah River. This spring closure likely benefits sturgeon as well. Seasonal restrictions are also placed on hopper dredging conducted offshore of Savannah Harbor in the shipping channel to protect sea turtles. To reduce the impacts of dredging on anadromous fish species, most of the Atlantic states impose work restrictions during sensitive time periods (spawning, migration, feeding) when anadromous fish are present.

Water Quality

Shortnose sturgeon rely on a variety of water quality parameters to successfully carry out their life functions. Low DO and the presence of contaminants modify the quality of sturgeon habitat and, in some cases, restrict the extent of suitable habitat for life functions. Secor (1995) noted a correlation between low abundances of sturgeon during this century and decreasing water quality caused by increased nutrient loading and increased spatial and temporal frequency of hypoxic (low oxygen) conditions.

Shortnose sturgeon do appear to become more resilient to low levels of DO as they age. Jenkins Jr. et al. (1993) exposed 11-330 day old shortnose sturgeon to a range of DO levels at a static temperature of 22.5°C (72.5°F) for 6 hours. DO concentrations of 2.5 mg/L killed 100% of 25-day-old fish, 96% of fish 32 days old, and 86% of fish 64 days old but only 12% of the fish older than 104 days (Jenkins Jr. et al. 1993). Jenkins Jr. et al. (1993) also reported young fish died at significantly higher rates for DO concentrations of 3.0 mg/L, while this concentration did not appear to adversely affect fish >77 days old. They also concluded that regardless of age, groups exposed to 2.0 mg/L died at significantly higher rates than the control groups (Jenkins Jr. et al. 1993).

Campbell and Goodman (2004) investigate the environmental impacts of shortnose sturgeon by considered the relationship between DO, salinity, and temperature. They conducted tests with hatchery-produced fish exposed to the ranges of DO, salinity, and temperature, similar to what might be expected in the southeastern United States coastal river–estuary interfaces during spring and summer. For 77-day-old fish, they determined the 50% mortality in 24 hours was likely when exposed to a combination of 2 parts per thousand salinity, a temperature of 25°C (77°F), and a DO level of 2.7 mg/L. In older fish (104-days-old), a 50% mortality rate in 24 hours occurred with DO concentrations of 2.2 mg/L at 22°C (71.6°F) and salinities of 4 parts per thousand (Campbell and Goodman 2004). However, even with relatively higher DO concentration (3.1 mg/L), Campbell and Goodman (2004) reported a 50% mortality rate in 24 hours for 100-day-old fish when temperature increased to of 30°C (86°F), even if the salinity decreased to 2 parts per thousand.

These studies highlight the concern regarding the high occurrence of low DO coupled with elevated temperatures in the river systems throughout the range of the shortnose sturgeon in the Southeast. For example, shallow water in many of the estuaries and rivers in North Carolina and South Carolina will reach temperatures nearing 30^{o}C in the summer months. Both low flow and high water temperatures can cause DO levels to drop to less than 3.0 mg/L. Sturgeon are more sensitive to low DO than other fish species (Niklitschek and Secor 2009a; Niklitschek and Secor 2009b), and low DO in combination with high temperature is particularly problematic.

Elevated levels of environmental contaminants, including chlorinated hydrocarbons, in several fish species are associated with reproductive impairment (Cameron et al. 1992; Longwell et al. 1992), reduced egg viability (Hansen 1985; Mac and Edsall 1991; von Westernhagen et al. 1981), and reduced survival of larval fish (Berlin et al. 1981; Giesy et al. 1986). Several characteristics of shortnose sturgeon (i.e., long life span, extended residence in estuarine habitats, benthic predator) predispose the species to long-term and repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants (Dadswell 1979). Chemicals and metals such as chlordane, dichlorodiphenyldichloroethylene, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later consumed by benthic feeders such as sturgeon or macroinvertebrates, and then work their way into the food web. Some of these compounds may affect physiological processes and impede a fish's ability to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing DO, altering pH, and altering other physical properties of the waterbody. Exposure to sufficient concentrations of these chemicals can cause lethal and sublethal effects such as: behavioral alterations, deformities, reduced growth, reduced fecundity, and reduced egg viability (Ruelle and Keenlyne 1993; USFWS 1993).

Water Quantity

Water allocation issues are a growing threat in the Southeast and exacerbate existing water quality problems. Taking water from one basin and transferring it to another fundamentally and irreversibly alters natural water flows in both the originating and receiving basins. This transfer can affect DO levels, temperature, and the ability of the basin of origin to assimilate pollutants (GWC 2006b). Water quality within the river systems in the range of the shortnose sturgeon is negatively affected by large water withdrawals. Known water withdrawals of over 240 million gallons per day are permitted from the Savannah River for power generation and municipal uses. However, permits for users withdrawing less than 100,000 gallons per day are not required, so actual water withdrawals from the Savannah River and other rivers within the range of the shortnose sturgeon are likely much higher. The removal of large amounts of water from the system alters flows, temperature, and DO. Water shortages and "water wars" are already occurring in the rivers occupied by the shortnose sturgeon and will likely be compounded in the future by human population growth and potentially by climate change.

Climate Change
Large-scale factors impacting riverine water quality and quantity that likely exacerbate habitat threats to shortnose sturgeon include drought, and intra- and inter-state water allocation. Changes in the climate are very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. For example, while annual precipitation in the Southeast has increased by 0.19 inches per decade since 1950 (NOAA 2018a), the southeastern United States has experienced several years of drought since 2007. During this time, Georgia and South Carolina experienced drought conditions that ranged from moderate to extreme. Between March 2007 and December 2008, 50-100% of the State of Georgia and the State of South Carolina experienced some level of drought ranging in intensity from "abnormally dry" to "exceptional," based on the drought intensity categories used by the U.S. Drought Monitor (National Drought Mitigation Center 2018). That drought was surpassed just a few years later. Both states again experienced "abnormally dry" to "exceptional" drought conditions across 50-100% of those states again from September 2010-March 2013, experienced "abnormally dry" to "exceptional" drought conditions https://droughtmonitor.unl.edu/Data/Timeseries.aspx (National Drought Mitigation Center 2018). While Georgia has periodically undergone periods of drought—there have been 6 periods of drought lasting from 2-7 years since 1903 (Barber and Stamey 2000)—drought frequency appears to be increasing (Ruhl 2003). Abnormally low stream flows can restrict access by sturgeon to habitat areas and exacerbate water quality issues such as water temperature, reduced DO, nutrient levels, and contaminants.

Long-term observations also confirm changes in temperature are occurring at a rapid rate. From 1895-2017, the average annual temperature in the Southeast has risen 0.1°F per decade. From 1950-2017, the increase triples to 0.3°F per decade per decade (NOAA 2018a). Aside from observation, climate modeling also projects future increases in temperatures in the Southeast. Table 35 summarizes the increases projected for the Southeast by the mid-century (2036–2065) and late-century (2071–2100). These are projections from the RCP model scenarios RCP8.5 and RCP4.5, used by the Intergovernmental Panel on Climate Change, relative to average from 1976–2005 (Hayhoe et al. 2017). RCPs make predictions based on changes, if any, in future greenhouse gas emissions. Specifically, they evaluate radiative forcing, or the amount of energy stored at the Earth's surface in watts/meters^2. As the amount of greenhouse gases increases in the atmosphere more energy is trapped, and the number of watts/m^2 increases. RCP2.6 and RCP8.5 represent the lowest and highest radiative scenarios, of 2.6 watts/meters^2 and 8.5 watts/meters^2, respectively. RCP4.5 and RCP6.0 assume intermediate levels of radiative forcing.

Table 35. Projected Temperature Increase in the Southeast Under Two Model Projections and Time Series (Hayhoe et al. 2017)

National Climate Assessment Region	RCP4.5 Mid-Century (2036–2065)	RCP8.5 Mid-Century (2036–2065)	RCP4.5 Late-Century (2071–2100)	RCP8.5 Late-Century (2071–2100)
Southeast	3.40°F (1.89°C)	4.30°F (2.39°C)	4.43°F (2.46°C)	7.72°F (4.29°C)

Shortnose sturgeon are already susceptible to reduced water quality resulting from dams, inputs of nutrients, contaminants from industrial activities and nonpoint sources, and interbasin transfers of water. The Intergovernmental Panel on Climate Change projects with high confidence that higher water temperatures and changes in extremes in the Southeast region, including floods and droughts, will affect water quality and exacerbate many forms of water pollution from sediments, nutrients, dissolved organic carbon, pathogens, pesticides, and salt, as well as thermal pollution, with possible negative impacts on ecosystems (Intergovernmental Panel on Climate Change 2007).

Sea-level rise is another consequence of climate change; it has already had significant impacts on coastal areas and these impacts are likely to increase. Since 1852, when the first topographic maps of the Southeastern United States were prepared, high tidal flood elevations have increased approximately 12 inches. During the 20th century, global sea level has increased 15 to 20 cm (NAST 2000). Sea level rise is also projected to extend areas of salinization of groundwater and estuaries. Some of the most populated areas of this region are low-lying; the threat of saltwater entering into this region’s aquifers with projected sea level rise is a concern (USGRG 2004). Saltwater intrusion will likely exacerbate existing water allocation issues, leading to an increase in reliance on interbasin water transfers to meet municipal water needs, further stressing water quality. Similarly, saltwater intrusion is likely to affect local ecosystems. Analysts attribute the forest decline in the Southeast to saltwater intrusion associated with sea level rise. Coastal forest losses will be even more severe if sea level rise accelerates as is expected as a result of global warming.

The effects of future climate change will vary greatly in diverse coastal regions for the United States. Warming is very likely to continue in the United States during the next 25 to 50 years, regardless of reduction in greenhouse gases, due to emissions that have already occurred (National Assessment Synthesis Team 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25 to 50 years, and it is possible that they will accelerate. A warmer and drier climate would reduce stream flows and increase water temperatures. Expected consequences would be a decrease in the amount of DO in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer, wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch et al. 2000).

Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the Southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer et al. 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change. Within 50 years, river basins that are impacted by dams or by extensive development, like the Savannah or Cooper River, will experience greater changes in discharge and water stress than unimpacted, free-flowing rivers (Palmer et al. 2008).

Dams, dredging, and poor water quality have already modified and restricted the extent of suitable habitat for shortnose sturgeon spawning and nursery habitat. Changes in water availability (depth and velocities) and water quality (temperature, salinity, DO, contaminants, etc.) in rivers and coastal waters inhabited by shortnose sturgeon resulting from climate change will further modify and restrict the extent of suitable habitat. Effects could be especially harmful since these populations have already been reduced to low numbers, potentially limiting their capacity for adaptation to changing environmental conditions (Belovsky 1987; Salwasser et al. 1984; Soulé 1987; Thomas 1990).

Bycatch
Overutilization of shortnose sturgeon from directed fishing caused initial severe declines in shortnose sturgeon populations in the Southeast, from which they have never rebounded. Further, continued collection of shortnose sturgeon as bycatch in commercial fisheries is an ongoing impact. Shortnose sturgeon are incidentally caught in state shad gillnet fisheries is occurring in the Ogeechee (NMFS 2010b) and Altamaha (Bahn et al. 2012) rivers. Shortnose sturgeon are sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum reproductive rates, and a large percentage of egg production occurs later in life. In addition, stress or injury to shortnose sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

As a wide-ranging anadromous species, shortnose sturgeon are subject to numerous federal (United States and Canadian), state, provincial, and interjurisdictional laws, regulations, and agencies' activities. While these mechanisms have addressed impacts to shortnose sturgeon through directed fisheries, there are currently no mechanisms in place to address the significant risk posed to shortnose sturgeon from commercial bycatch. Though statutory and regulatory mechanisms exist that authorize reducing the impact of dams on riverine and anadromous species, such as shortnose sturgeon, and their habitat, these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Further, water quality continues to be a problem in the historical spawning rivers along the Atlantic coast, even with existing controls on some pollution sources. Current regulatory authorities are not necessarily effective in controlling water allocation issues (e.g., no restrictions on interbasin water transfers in South Carolina, the lack of ability to regulate non-point source pollution).

Stochastic Events

Stochastic events, such as hurricanes, are common throughout the range of shortnose sturgeon. These events are unpredictable and their effect on the survival and recovery of the species in unknown; however, they have the potential to impede the survival and recovery directly if animals die as a result of them, or indirectly if habitat, is damaged as a result of these disturbances. For example, in 2018, flooding from Hurricane Florence flushed significant amounts of organic matter into rivers supporting sturgeon. The DO levels in those rivers dropped so low (i.e., 0.2 mg/L) that thousands of fish suffocated, including multiple sturgeon.

4.1.4 Giant manta

NMFS listed the giant manta ray (*Manta birostris*) as threatened under the ESA (83 FR 2916, Publication Date January 22, 2018) and determined that the designation of critical habitat is not prudent on (84 FR 66652, Publication Date December 5, 2019). On December 4, 2019, NMFS published a recovery outline for the giant manta ray (NMFS 2019a), which serves as an interim guidance to direct recovery efforts for giant manta ray.

4.1.4.1 Species Description and Distribution

The giant manta ray is the largest living ray, with a wingspan reaching a width of up to 7 m (23 ft), and an average size between 4-5 m (15-16.5 ft). The giant manta ray is recognized by its large diamond-shaped body with elongated wing-like pectoral fins, ventrally placed gill slits, laterally placed eyes, and wide terminal mouth. In front of the mouth, it has 2 structures called cephalic lobes that extend and help to introduce water into the mouth for feeding activities (making them the only vertebrate animals with 3 paired appendages). Giant manta rays have 2 distinct color types: chevron (mostly black back dorsal side and white ventral side) and black (almost completely black on both ventral and dorsal sides). Most of the chevron variants have a black dorsal surface and a white ventral surface with distinct patterns on the underside that can be used to identify individuals (Miller and Klimovich 2017). There are bright white shoulder markings on the dorsal side that form 2 mirror image right-angle triangles, creating a T-shape on the upper shoulders.

The giant manta ray can be found in all ocean basins. In terms of range, within the Northern hemisphere, the species has been documented as far north as southern California and New Jersey on the United States west and east coasts, respectively, and Mutsu Bay, Aomori, Japan, the Sinai Peninsula and Arabian Sea, Egypt, and the Azores Islands (CITES 2013; Gudger 1922;

Kashiwagi et al. 2010; Moore 2012). In the Southern Hemisphere, the species occurs as far south as Peru, Uruguay, South Africa, New Zealand and French Polynesia (CITES 2013; Mourier 2012). Within its range, the giant manta ray inhabits tropical, subtropical, and temperate bodies of water and is commonly found offshore, in oceanic waters, and near productive coastlines (Figure 31) (Kashiwagi et al. 2011; Marshall et al. 2009).

Figure 31. The Extent of Occurrence (dark blue) and Area of Occupancy (light blue) based on species distribution (Lawson et al. 2017).

4.1.4.2 Life History Information

Giant manta rays make seasonal long-distance migrations, aggregate in certain areas and remain resident, or aggregate seasonally (Dewar et al. 2008; Girondot et al. 2015; Graham et al. 2012; Stewart et al. 2016). The giant manta ray is a seasonal visitor along productive coastlines with regular upwelling, in oceanic island groups, and at offshore pinnacles and seamounts. The timing of these visits varies by region and seems to correspond with the movement of zooplankton, current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior. They have also been observed in estuarine waters near oceanic inlets, with use of these waters as potential nursery grounds (Adams and Amesbury 1998; Medeiros et al. 2015; Milessi and Oddone 2003) J. Pate, Florida Manta Project, unpublished data).

Giant manta rays are known to aggregate in various locations around the world in groups usually ranging from 100-1,000 (Graham et al. 2012; Notarbartolo di Sciara and Hillyer 1989; Venables 2013). These sites function as feeding sites, cleaning stations, or sites where courtship interactions take place (Graham et al. 2012; Heinrichs et al. 2011; Venables 2013). The appearance of giant manta rays in these locations is generally predictable. For example, food availability due to high productivity events tends to play a significant role in feeding site aggregations (Heinrichs et al. 2011; Notarbartolo di Sciara and Hillyer 1989). Giant manta rays have also been shown to return to a preferred site of feeding or cleaning over extended periods of time (Dewar et al. 2008; Graham et al. 2012; Medeiros et al. 2015). In addition, giant and reef manta rays in Keauhou and Ho”ona Bays in Hawaii, appear to exhibit learned behavior. These manta rays learned to associate artificially lighting with high plankton concertation (primary food source) and shifted foraging strategies to include sites that had artificially lighting at night (Clark 2010). While little is known about giant manta ray aggregation sites, the Flower Garden Banks National Marine Sanctuary and the surrounding region might represent the first documented nursery habitat for giant manta ray (Stewart et al. 2018). Stewart et al. (2018) found that the Flower Garden Banks National Marine Sanctuary provides nursery habitat for juvenile giant manta rays because small age classes have been observed consistently across years at both the population and individual level. The Flower Garden Banks National Marine Sanctuary may be an optimal nursery ground because of its location near the edge of the continental shelf and proximity to abundant pelagic food resources. In addition, small juveniles are frequently observed along a portion of Florida's east coast, indicating that this area may also function as a nursery ground for juvenile giant manta rays. Since directed visual surveys began in 2016, juvenile giant manta rays are regularly observed in the shallow waters (less than 5 m depth) from Jupiter Inlet to Boynton Beach Inlet (J Pate, Florida Manta Project, unpublished data). However, the extent of this purported nursery ground is unknown as the survey area is limited to a relatively narrow geographic area along Florida's east coast.

The giant manta ray appears to exhibit a high degree of plasticity in terms of its use of depths within its habitat. Tagging studies have shown that the giant manta rays conduct night descents from 200-450m depths (Rubin et al. 2008; Stewart et al. 2016) and are capable of diving to depths exceeding 1,000 m (A. Marshall et al. unpublished data 2011, cited in Marshall et al. (2011)). Stewart et al. (2016) found diving behavior may be influenced by season, and more specifically, shifts in prey location associated with the thermocline, with tagged giant manta rays (n=4) observed spending a greater proportion of time at the surface from April to June and in deeper waters from August to September. Overall, studies indicate that giant manta rays have a more complex depth profile of their foraging habitat than previously thought, and may actually be supplementing their diet with the observed opportunistic feeding in near-surface waters (Burgess et al. 2016; Couturier et al. 2013).

Giant manta rays primarily feed on planktonic organisms such as euphausiids, copepods, mysids, decapod larvae and shrimp, but some studies have noted their consumption of small and moderately sized fishes (Miller and Klimovich 2017). While it was previously assumed, based on field observations, that giant manta rays feed predominantly during the day on surface zooplankton, results from recent studies (Burgess et al. 2016; Couturier et al. 2013) indicate that these feeding events are not an important source of the dietary intake. When feeding, giant manta rays hold their cephalic lobes in an "O" shape and open their mouth wide, which creates a funnel that pushes water and prey through their mouth and over their gill rakers. They use many different types of feeding strategies, such as barrel rolling (doing somersaults repeatedly) and creating feeding chains with other mantas to maximize prey intake.

The giant manta ray is viviparous (i.e., gives birth to live young). They are slow to mature and have very low fecundity and typically give birth to only one pup every 2 to 3 years. Gestation lasts approximately 10-14 months. Females are only able to produce between 5 and 15 pups in a lifetime (CITES 2013; Miller and Klimovich 2017). The giant manta ray has one of the lowest maximum population growth rates of all elasmobranchs (Dulvy et al. 2014; Miller and Klimovich 2017). The giant manta rays generation time (based on *M. alfredi* life history parameters) is estimated to be 25 years (Miller and Klimovich 2017).

Although giant manta rays have been reported to live at least 40 years, not much is known about their growth and development. Maturity is thought to occur between 8-10 years of age (Miller and Klimovich 2017). Males are estimated to mature at around 3.8 m disc width (slightly smaller than females) and females at 4.5 m disc width (Rambahiniarison et al. 2018).

4.1.4.3 Status and Population Dynamics

There are no current or historical estimates of global abundance of giant manta rays, with most estimates of subpopulations based on anecdotal observations. The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 2013) found that only ten populations of giant manta rays had been actively studied, 25 other aggregations have been anecdotally identified, all other sightings are rare, and the total global population may be small. Subpopulation abundance estimates range between 42 and 1,500 individuals, but are anecdotal and subject to bias (Miller and Klimovich 2017). The largest subpopulations and records of individuals come from the Indo-Pacific and eastern Pacific. Ecuador is thought to be home to the largest identified population (n=1,500) of giant manta rays in the world, with large aggregation sites within the waters of the Machalilla National Park and the Galapagos Marine Reserve (Hearn et al. 2014). Within the Indian Ocean, numbers of giant manta rays identified through citizen science in Thailand's waters (primarily on the west coast, off Khao Lak and Koh Lanta) was 288 in 2016. These numbers reportedly surpass the estimate of identified giant mantas in Mozambique (n=254), possibly indicating that Thailand may be home to the largest aggregation of giant manta rays within the Indian Ocean (MantaMatcher 2016). Miller and Klimovich (2017) concluded that giant manta rays are at risk throughout a significant portion of their range, due in large part to the observed declines in the Indo-Pacific. There have been decreases in landings of up to 95% in the Indo-Pacific, although similar declines have not been observed in areas with other subpopulations, such as Mozambique and Ecuador. In the U.S. Atlantic, the giant manta rays appear to have a seasonal pattern of occurrence along the east coast of Florida, showing up with greater frequencies (and in greater numbers) in the spring and summer months (84 FR 66652; Publication Date December 5, 2019). Available sightings data indicates the seasonal visitation of manta rays to Florida's inshore waters, possible juvenile habitat, and possible residency. The numbers, location, and peak timing of the manta rays to this area varies by year (H. Webb unpublished data). In 2015, aerial survey conducted by the Georgia Aquarium peaked at 1,144 manta ray sighted in the inshore waters of northeast Florida, but with notable decline in manta rays observed in the study area since 2015 (H. Webb unpublished data). In addition, juvenile giant manta rays have also been regularly observed inshore off the southeast Florida. Since 2016, researchers with the Marine Megafauna Foundation have been conducting annual surveys along a small transect off Palm Beach, Florida, between Jupiter Inlet and Boynton Beach Inlet (~44 km, 24 nautical miles) (J. Pate, MMF, pers. comm. to M. Miller, NMFS OPR, 2018). Results from these surveys indicate that juvenile manta rays are present in these waters for the majority of the year (observations span from May to December), with re-sightings data that suggest some manta rays may remain in the area for extended periods of time or return in subsequent years (J. Pate unpublished data). In the Gulf of Mexico, within the Flower Garden Banks National Marine Sanctuary, 95 unique individuals have been recorded between 1982 and 2017 (Stewart et al. 2018).

4.1.4.4 Threats

The giant manta ray faces many threats, including fisheries interactions, environmental contaminants (microplastics, marine debris, petroleum products, etc.), vessel strikes, entanglement, and global climate change. Overall, the predictable nature of their appearances, combined with slow swimming speed, large size, and lack of fear towards humans, may increase their vulnerability to threats (Convention on Migratory Species 2014; O'Malley et al. 2013). The ESA status review determined that the greatest threat to the species results from fisheries related mortality (Miller and Klimovich 2017); (83 FR 2916, Publication Date January 22, 2018).

Commercial Harvest and Fisheries Bycatch
Commercial harvest and incidental bycatch in fisheries is cited as the primary cause for the decline in the giant manta ray and threat to future recovery (Miller and Klimovich 2017). We anticipate that these threats will continue to affect the rate of recovery of the giant manta ray. Worldwide giant manta ray catches have been recorded in at least 30 large and small-scale fisheries covering 25 countries (Lawson et al. 2016). Demand for the gills of giant manta rays and other mobula rays has risen dramatically in Asian markets. With this expansion of the international gill raker market and increasing demand for manta ray products, estimated harvest of giant manta rays, particularly in many portions of the Indo-Pacific, frequently exceeds numbers of identified individuals in those areas and are accompanied by observed declines in sightings and landings of the species of up to 95% (Miller and Klimovich 2017). In the Indian Ocean, manta rays (primarily giant manta rays) are mainly caught as bycatch in purse seine and gillnet fisheries (Oliver et al. 2015). In the western Indian Ocean, data from the pelagic tuna purse seine fishery suggests that giant manta and mobula rays, together, are an insignificant portion of the bycatch, comprising less than 1% of the total non-tuna bycatch per year (Chassot et al. 2008; Romanov 2002). In the U.S., bycatch of giant manta rays has been recorded in the coastal migratory pelagic gillnet, gulf reef fish bottom longline, Atlantic shark gillnet, pelagic longline, pelagic bottom longline, and trawl fisheries. Incidental capture of giant manta ray is also a rare occurrence in the elasmobranch catch within U.S. Atlantic and Gulf of Mexico, with the majority that are caught released alive. In addition to directed harvest and bycatch in commercial fisheries, the giant manta ray is incidentally captured by recreational fishers using vertical line (i.e., handline, bandit gear, and rod-and-reel). Researchers frequently report giant manta rays having evidence of recreational gear interactions along the east coast of Florida (i.e., manta rays have embedded fishing hooks with attached trailing monofilament line) (J. Pate, Florida Manta Project, unpublished data). Internet searches also document recreational interactions with giant manta rays. For example, recreational fishers will search for giant manta rays while targeting cobia, as cobia often accompany giant manta rays (anglers will cast at manta rays in an effort to hook cobia). In addition, giant manta rays are commonly observed swimming near or underneath public fishing piers where they may become foul-hooked. The current threat of mortality associated with recreational fisheries is expected to be low, given that we have no reports of recreational fishers retaining giant manta ray. However, bycatch in recreational fisheries remains a potential threat to the species.

Vessel Strike
Vessel strikes can injure or kill giant manta rays, decreasing fitness or contributing to non-natural mortality (Couturier et al. 2012; Deakos et al. 2011). Giant manta rays can be frequently observed traveling just below the surface and will often approach or show little fear toward

humans or vessels (Coles 1916a), which can also make them extremely vulnerable to vessel strikes (Deakos 2010). Five giant manta rays were reported to have been struck by vessels from 2016 through 2018; individuals had injuries (i.e., fresh or healed dorsal surface propeller scars) consistent with a vessel strike. These interactions were observed by researchers conducting surveys from Boynton Beach to Jupiter, Florida (J. Pate, Florida Manta Project, unpublished data). The giant manta ray is frequently observed in nearshore coastal waters and feeding at inlets along the east coast of Florida. As vessel traffic is concentrated in and around inlets and nearshore waters, this overlap exposes the giant manta ray in these locations to an increased likelihood of potential vessel strike injury. Yet, few instances of confirmed or suspected mortalities of giant manta ray attributed to vessel strike injury (e.g., via strandings) have been documented. This lack of documented mortalities could also be the result of other factors that influence carcass detection (i.e., wind, currents, scavenging, decomposition etc.).

Microplastics

Filter-feeding megafauna are particularly susceptible to high levels of microplastic ingestion and exposure to associated toxins due to their feeding strategies, target prey, and, for most, habitat overlap with microplastic pollution hotspots (Germanov et al. 2019). Giant manta rays are filter feeders, and, therefore can ingest microplastics directly from polluted water or indirectly through-contaminated planktonic prey (Miller and Klimovich 2017). The effects of ingesting indigestible particles include blocking adequate nutrient absorption and causing mechanical damage to the digestive tract. Microplastics can also harbor high levels of toxins and persistent organic pollutants, and introduce these toxins to organisms via ingestion. These toxins can bioaccumulate over decades in long-lived filter feeders, leading to a disruption of biological processes (e.g., endocrine disruption), and potentially altering reproductive fitness (Germanov et al. 2019). Jambeck et al. (2015) found that the Western and Indo- Pacific regions are responsible for the majority of plastic waste. These areas also happen to overlap with some of the largest known aggregations of giant manta rays. For example, in Thailand, where recent sightings data have identified over 288 giant manta rays (MantaMatcher 2016), mismanaged plastic waste is estimated to be on the order of 1.03 million tonnes annually, with up to 40% of this entering the marine environment (Jambeck et al. 2015). Approximately 1.6 million tonnes of mismanaged plastic waste is being disposed of in Sri Lanka, again with up to 40% entering the marine environment (Jambeck et al. 2015), potentially polluting the habitat used by the nearby Maldives aggregation of manta rays. While the ingestion of plastics is likely to negatively affect the health of the species, the levels of microplastics in manta ray feeding grounds and frequency of ingestion are presently being studied to evaluate the impact on these species (Germanov et al. 2019).

Mooring and Anchor Lines

Mooring and boat anchor line entanglement may also wound giant manta rays or cause them to drown (Deakos et al. 2011; Heinrichs et al. 2011). There are numerous anecdotal reports of giant manta rays becoming entangled in mooring and anchor lines (C. Horn, NMFS, unpublished data), as well as documented interactions encountered by other species of manta rays (C. Horn, NMFS, unpublished data). For example, although a rare occurrence, reef manta rays on occasion entangle themselves in anchor and mooring lines. Deakos (2010) suggested that manta rays become entangled when the line makes contact with the front of the head between the cephalic lobes, the animal’s reflex response is to close the cephalic lobes, thereby trapping the rope between the cephalic lobes, entangling the manta ray as the animal begins to roll in an attempt to free itself. In Hawaii, on at least 2 occasions, a reef manta ray was reported to have died after entangling in a mooring line (A. Cummins, pers. comm. 2007, K. Osada, pers. comm. 2009; cited in Deakos (2011)). In Maui, Hawaii, Deakos et al. (2011) observed that 1 out of 10 reef manta rays had an amputated or disfigured non-functioning cephalic lobe, likely a result of line entanglement. Mobulid researchers indicate that entanglements may significantly affect the manta rays fitness (Braun et al. 2015; Convention on Migratory Species 2014; Couturier et al. 2012; Deakos et al. 2011; Germanov and Marshall 2014; Heinrichs et al. 2011). However, there is very little quantitative information on the frequency of these occurrences and no information on the impact of these injuries on the overall health of the species.

Climate Change Effects
Because giant manta rays are migratory and considered ecologically flexible (e.g., low habitat specificity), they may be less vulnerable to the impacts of climate change compared to other sharks and rays (Chin et al. 2010). However, as giant manta rays frequently rely on coral reef habitat for important life history functions (e.g., feeding, cleaning) and depend on planktonic food resources for nourishment, both of which are highly sensitive to environmental changes (Brainard et al. 2011; Guinder and Molinero 2013), climate change is likely to have an impact on their distribution and behavior. Coral reef degradation from anthropogenic causes, particularly climate change, is projected to increase through the future. Specifically, annual, globally averaged surface ocean temperatures are projected to increase by approximately 0.7 °C by 2030 and 1.4 °C by 2060 compared to the 1986-2005 average (Intergovernmental Panel on Climate Change 2013), with the latest climate models predicting annual coral bleaching for almost all reefs by 2050 (Heron et al. 2016). Declines in coral cover have been shown to result in changes in coral reef fish communities (Jones et al. 2004) (Graham et al. 2008). Therefore, the projected increase in coral habitat degradation may potentially lead to a decrease in the abundance of fish that clean giant manta rays (e.g., *Labroides* spp., *Thalassoma* spp., and *Chaetodon* spp.) and an overall reduction in the number of cleaning stations available to manta rays within these habitats. Decreased access to cleaning stations may negatively affect the fitness of giant manta rays by hindering their ability to reduce parasitic loads and dead tissue, which could lead to increases in diseases and declines in reproductive fitness and survival rates.

Changes in climate and oceanographic conditions, such as acidification, are also known to affect zooplankton structure (size, composition, and diversity), phenology, and distribution (Guinder and Molinero 2013). As such, the migration paths and locations of both resident and seasonal aggregations of giant manta rays, which depend on these animals for food, may similarly be altered (Couturier et al. 2012). As research to understand the exact impacts of climate change on marine phytoplankton and zooplankton communities is still ongoing, the severity of this threat has yet to be fully determined (Miller and Klimovich 2017).

4.1.5 Smalltooth sawfish

The U.S. DPS of smalltooth sawfish was listed as endangered under the ESA effective May 1, 2003 (68 FR 15674; Publication Date April 1, 2003).

4.1.5.1 Species Description and Distribution

The smalltooth sawfish is a tropical marine and estuarine elasmobranch. It is a batoid with a long, narrow, flattened, rostral blade (rostrum) lined with a series of transverse teeth along either edge. In general, smalltooth sawfish inhabit shallow coastal waters of the Atlantic Ocean (Dulvy et al. 2016) and feed on a variety of fish (e.g., mullet, jacks, and ladyfish) (Poulakis et al. 2017; Simpfendorfer 2001).

Although this species is reported throughout the tropical Atlantic, NMFS identified smalltooth sawfish from the Southeast United States as a DPS, due to the physical isolation of this population from others, the differences in international management of the species, and the significance of the U.S. population in relation to the global range of the species (68 FR15674, Publication Date April 1, 2003). Within the United States, smalltooth sawfish have historically been captured in estuarine and coastal waters from North Carolina southward through Texas, although peninsular Florida has been the region of the United States with the largest number of recorded captures (NMFS 2018e). Recent records indicate there is a resident reproducing population of smalltooth sawfish in south and southwest Florida from Charlotte Harbor through the Florida Keys, which is also the last U.S. stronghold for the species (Poulakis and Seitz 2004; Seitz and Poulakis 2002; Simpfendorfer and Wiley 2005). Water temperatures (no lower than 8-12°C) and the availability of appropriate coastal habitat (shallow, euryhaline waters and red mangroves) are the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic. Most specimens captured along the Atlantic coast north of Florida are large juveniles or adults (over 10 ft) that likely represent seasonal migrants, wanderers, or colonizers from a historical Florida core population to the south, rather than being members of a continuous, even-density population (Bigelow and Schroeder 1953b).

4.1.5.2 Life History Information

Smalltooth sawfish mate in the spring and early summer (Grubbs unpublished data; Poulakis unpublished data). Fertilization is internal and females give birth to live young. Evidence suggests a gestation period of approximately 12 months and females produce litters of 7-14 young (Feldheim et al. 2017)(Gelsleichter unpub. data). Females have a biennial reproductive cycle (Feldheim et al. 2017) and parturition (act of giving birth) occurs nearly year round though peaking in spring and early summer (March – July) (Poulakis et al. 2011a)(Carlson unpublished data). Smalltooth sawfish are approximately 26-31 in (64-80 cm) at birth (Bethea et al. 2012; Poulakis et al. 2011a) and may grow to a maximum length of approximately 16 ft (500 cm) (Grubbs unpublished data (Brame et al. 2019). Simpfendorfer et al. (2008) report rapid juvenile growth for smalltooth sawfish for the first 2 years after birth, with stretched total length increasing by an average of 25-33 in (65-85 cm) in the first year and an average of 19-27 in (48-68 cm) in the second year. Uncertainty remains in estimating post-juvenile growth rates and age at maturity; yet, recent advances indicate maturity at 7-11 years (Carlson and Simpfendorfer 2015) at lengths of approximately 340 cm for males and 350-370 cm for females (Gelsleichter unpub data).

There are distinct differences in habitat use based on life history stage as the species shifts use through ontogeny. Juvenile smalltooth sawfish less than 220 cm, inhabit the shallow euryhaline waters (i.e., variable salinity) of estuaries and can be found in sheltered bays, dredged canals, along banks and sandbars, and in rivers (NMFS 2000). These juveniles are often closely associated with muddy or sandy substrates, and shorelines containing red mangroves, *Rhizophora mangle* (Hollensead et al. 2016; Hollensead et al. 2018; Poulakis et al. 2011a; Poulakis et al. 2013; Simpfendorfer 2001; Simpfendorfer 2003; Simpfendorfer et al. 2010). (Simpfendorfer et al. 2010) indicated the smallest juveniles (young-of-the-year juveniles measuring < 100 cm in length) generally used the shallowest water (depths less than 0.5 m [1.64 ft]), had small home ranges (4,264-4,557 square meters [m^2]), and exhibited high levels of site fidelity. Although small juveniles exhibit high levels of site fidelity for specific nursery habitats for periods of time lasting up to 3 months (Wiley and Simpfendorfer 2007), they do undergo small movements coinciding with changing tidal stages. These movements often involve moving from shallow sandbars at low tide to within red mangrove prop roots at higher tides (Simpfendorfer et al. 2010)—behavior likely to reduce the risk of predation (Simpfendorfer 2006). As juveniles increase in size, they begin to expand their home ranges (Simpfendorfer et al. 2010; Simpfendorfer et al. 2011), eventually moving to more offshore habitats where they likely feed on larger prey as they continue to mature.

Researchers have identified several areas within the Charlotte Harbor Estuary that are disproportionately more important to juvenile smalltooth sawfish, based on intra- or inter-annual (within or between year) capture rates during random sampling events within the estuary(Poulakis et al. 2011a; Poulakis 2012). These high-use areas were termed “hotspots” and also correspond with areas where public encounters are most frequently reported. Use of these “hotspots” can vary within and among years based on the amount and timing of freshwater inflow. Juvenile smalltooth sawfish use hotspots further upriver during high salinity conditions (drought) and areas closer to the mouth of the Caloosahatchee River during times of high freshwater inflow (Poulakis et al. 2011a). At this time, researchers are unsure what specific biotic or abiotic factors influence this habitat use, but they believe a variety of conditions in addition to salinity, such as temperature, DO, water depth, shoreline vegetation, and food availability, may influence habitat selection (Poulakis et al. 2011a).

The juvenile “hotspots” may be of further significance following the findings of female philopatry (Feldheim et al. 2017). More specifically, Feldheim et al. (2017) found that female sawfish return to the same parturition (birthing) sites over multiple years (parturition site fidelity). NMFS expects that these parturition sites align closely with the juvenile “hotspots” given the high fidelity shown by the smallest size/age classes of sawfish to specific nursery areas. Therefore, disturbance of these nursery areas could have wide-ranging effects on the sawfish population if it were to disrupt future parturition.

While adult smalltooth sawfish may also use the estuarine habitats used by juveniles, they are commonly observed in deeper waters along the coasts. Poulakis and Seitz (2004) noted that nearly half of the encounters with adult-sized smalltooth sawfish in Florida Bay and the Florida Keys occurred in depths from 200-400 ft (70-122 m) of water. Similarly, Simpfendorfer and Wiley (2005) reported encounters in deeper waters off the Florida Keys, and observations from both commercial longline fishing vessels and fishery-independent sampling in the Florida Straits report large smalltooth sawfish in depths up to 130 ft (~40 m)(ISED 2014). Yet, current field studies show adult smalltooth sawfish also use shallow estuarine habitats within Florida Bay and the Everglades (Grubbs unpub. data). Further, NMFS expects that females return to shallow estuaries during parturition (when adult females return to shallow estuaries to give birth).

4.1.5.3 Status and Population Dynamics

Based on the contraction of the species' geographic range, we expect that the population to be a fraction of its historical size. However, few long-term abundance data exist for the smalltooth sawfish, making it very difficult to estimate the current population size. Despite the lack of scientific data, recent encounters with young-of-the-year, older juveniles, and sexually mature smalltooth sawfish indicate that the U.S. population is currently reproducing (Feldheim et al. 2017; Seitz and Poulakis 2002; Simpfendorfer 2003). The abundance of juveniles publically encountered by anglers and boaters, including very small individuals, suggests that the population remains viable (Simpfendorfer and Wiley 2004), and data analyzed from Everglades National Park as part of an established fisheries-dependent monitoring program (angler interviews) indicated a slightly increasing trend in juvenile abundance within the park over the past decade (Carlson and Osborne 2012b; Carlson et al. 2007). Similarly, preliminary results of juvenile smalltooth sawfish sampling programs in both Everglades National Park and Charlotte Harbor indicate the juvenile population is at least stable and possibly increasing (Poulakis unpublished data, Carlson unpublished data).

Using a demographic approach and life history data for smalltooth sawfish and similar species from the literature, (Simpfendorfer 2000) estimated intrinsic rates of natural population increase for the species at 0.08-0.13 per year and population doubling times from 5.4-8.5 years. These low intrinsic rates[52] of population increase, suggest that the species is particularly vulnerable to excessive mortality and rapid population declines, after which recovery may take decades. Carlson and Simpfendorfer (2015) constructed an age-structured Leslie matrix model for the U.S. population of smalltooth sawfish, using updated life history information, to determine the species' ability to recover under scenarios of variable life history inputs and the effects of bycatch mortality and catastrophes. As expected, population growth was highest (λ=1.237 year-1) when age-at-maturity was 7 year and decreased to 1.150 year-1 when age-at-maturity was 11 year. Despite a high level of variability throughout the model runs, in the absence of fishing mortality or catastrophic climate effects, the population grew at a relatively rapid rate approaching carrying capacity in 40 years when the initial population was set at 2250 females or 50 years with an initial population of 600 females. Carlson and Simpfendorfer (2015) concluded that smalltooth sawfish in U.S. waters appear to have the ability to recover within the foreseeable future based on a model relying upon optimistic estimates of population size, lower age-at-maturity and the lower level of fisheries-related mortality. Another analysis was less optimistic based on lower estimates of breeding females in the Caloosahatchee River nursery (Chapman unpublished data). Assuming similar numbers of females among the 5 known nurseries, that study would suggest an initial breeding population of only 140-390 females, essentially half of the initial population considered by Carlson and Simpfendorfer (2015). A smaller initial breeding population would extend the time to reach carrying capacity.

[52] The rate at which a population increases in size if there are no density-dependent forces regulating the population

4.1.5.4 Threats

Past literature indicates smalltooth sawfish were once abundant along both coasts of Florida and quite common along the shores of Texas and the northern Gulf coast (NMFS 2010d) and citations therein). Based on recent comparisons with these historical reports, the U.S. DPS of smalltooth sawfish has declined over the past century(Simpfendorfer 2001; Simpfendorfer 2002). The decline in smalltooth sawfish abundance has been attributed to several factors including bycatch mortality in fisheries, habitat loss, and life history limitations of the species(NMFS 2010d).

4.1.5.5 Bycatch Mortality

Bycatch mortality is cited as the primary cause for the decline in smalltooth sawfish in the United States (NMFS 2010d). While there has never been a large-scale directed fishery, smalltooth sawfish easily become entangled in fishing gears (gill nets, otter trawls, trammel nets, and seines) directed at other commercial species, often resulting in serious injury or death (NMFS 2009c). This has historically been reported in Florida (Snelson and Williams 1981), Louisiana (Simpfendorfer 2002), and Texas (Baughman 1943). For instance, one fisherman interviewed by Evermann and Bean (1897) reported taking an estimated 300 smalltooth sawfish in just one netting season in the Indian River Lagoon, Florida. In another example, smalltooth sawfish landings data gathered by Louisiana shrimp trawlers from 1945-1978, which contained both landings data and crude information on effort (number of vessels, vessel tonnage, number of gear units), indicated declines in smalltooth sawfish landings from a high of 34,900 lbs in 1949 to less than 1,500 lbs in most years after 1967. The Florida net ban passed in 1995 has led to a reduction in the number of smalltooth sawfish incidentally captured, "…by prohibiting the use of gill and other entangling nets in all Florida waters, and prohibiting the use of other nets larger than 500 ft^2 in mesh area in nearshore and inshore Florida waters"[53] (FLA. CONST. art. X, § 16). However, the threat of bycatch currently remains in commercial fisheries (e.g., South Atlantic shrimp fishery, Gulf of Mexico shrimp fishery, federal shark fisheries of the South Atlantic, and the Gulf of Mexico reef fish fishery), though anecdotal information collected by NMFS port agents suggest smalltooth sawfish captures are now rare.

In addition to incidental bycatch in commercial fisheries, smalltooth sawfish have historically been and continue to be captured by recreational anglers. Encounter data (ISED 2014) and past research (Caldwell 1990) document that rostra are sometimes removed from smalltooth sawfish caught by recreational anglers, thereby reducing their chances of survival. While the current threat of mortality associated with recreational fisheries is expected to be low given that possession of the species in Florida has been prohibited since 1992, bycatch in recreational fisheries remains a potential threat to the species.

Habitat Loss

Modification and loss of smalltooth sawfish habitat, especially nursery habitat, is another contributing factor in the decline of the species. Activities such as agricultural and urban

[53] "nearshore and inshore Florida waters" means all Florida waters inside a line 3 miles seaward of the coastline along the Gulf of Mexico and inside a line 1 mile seaward of the coastline along the Atlantic Ocean.

development, commercial activities, dredge-and-fill operations, boating, erosion, and diversions of freshwater runoff contribute to these losses (SAFMC 1998). Large areas of coastal habitat were modified or lost between the mid-1970s and mid-1980s within the United States (Dahl and Johnson 1991). Since then, rates of loss have decreased, but habitat loss continues. From 1998-2004, approximately 64,560 acres of coastal wetlands were lost along the Atlantic and Gulf coasts of the United States, of which approximately 2,450 acres were intertidal wetlands consisting of mangroves or other estuarine shrubs (Steadman and Dahl 2008). Further, Orlando et al. (1994) analyzed 18 major southeastern estuaries and recorded over 703 miles of navigation channels and 9,844 miles of shoreline with modifications. In Florida, coastal development often involves the removal of mangroves and the armoring of shorelines through seawall construction. Changes to the natural freshwater flows into estuarine and marine waters through construction of canals and other water control devices have had other impacts: altered the temperature, salinity, and nutrient regimes; reduced both wetlands and submerged aquatic vegetation; and degraded vast areas of coastal habitat utilized by smalltooth sawfish (Gilmore 1995; Reddering 1988; Whitfield and Bruton 1989). While these modifications of habitat are not the primary reason for the decline of smalltooth sawfish abundance, it is likely a contributing factor and almost certainly hampers the recovery of the species. Juvenile sawfish and their nursery habitats are particularly likely to be affected by these kinds of habitat losses or alternations, due to their affinity for shallow, estuarine systems. (Prohaska et al. 2018a); Prohaska et al. (2018b) showed that juvenile smalltooth sawfish within the anthropogenically altered Charlotte Harbor estuary have higher metabolic stress compared to those collected from more pristine nurseries in the Everglades. Although many forms of habitat modification are currently regulated, some permitted direct and/or indirect damage to habitat from increased urbanization still occurs and is expected to continue to threaten survival and recovery of the species in the future.

Life History Limitations
The smalltooth sawfish is also limited by its life history characteristics as a relatively slow-growing, late-maturing, and long-lived species. Animals using this life history strategy are usually successful in maintaining small, persistent population sizes in constant environments, but are particularly vulnerable to increases in mortality or rapid environmental change (NMFS 2000). The combined characteristics of this life history strategy result in a very low intrinsic rate of population increase (Musick (editor) 1999) that make it slow to recover from any significant population decline (Simpfendorfer 2000).

Stochastic Events
Although stochastic events such as aperiodic extreme weather and harmful algal blooms are expected to affect smalltooth, we are currently unsure of their impact. A strong and prolonged cold weather event in January 2010 resulted in the mortality of at least 15 juvenile and 1 adult sawfish (Poulakis et al. 2011a; Scharer et al. 2012), and led to far fewer catches in directed research throughout the remainder of the year (Bethea et al. 2011). Another less severe cold front in 2011 did not result in any known mortality but did alter the typical habitat use patterns of juvenile sawfish within the Caloosahatchee River. Since surveys began, 2 hurricanes have made direct landfall within the core range of U.S. sawfish. While these storms denuded mangroves along the shoreline and created hypoxic water conditions, we are unaware of any direct effects to sawfish. Just prior to the passage of the most recent hurricane (Hurricane Irma), acoustically tagged sawfish moved away from their normal shallow nurseries and then returned within a few

days (Poulakis unpublished data; Carlson unpublished data). Harmful algal blooms have occurred within the core range of smalltooth sawfish and affected a variety of fauna including sea turtles, fish, and marine mammals, but to date no sawfish mortalities have been reported.

Current Threats
The 3 major factors that led to the current status of the U.S. DPS of smalltooth sawfish – bycatch mortality, habitat loss, and life history limitations – continue to be the greatest threats today. All the same, other threats such as the illegal commercial trade of smalltooth sawfish or their body parts, predation, and marine pollution and debris may also affect the population and recovery of smalltooth sawfish on smaller scales (NMFS 2010d). We anticipate that all of these threats will continue to affect the rate of recovery for the U.S. DPS of smalltooth sawfish.

In addition to the anthropogenic effects mentioned previously, changes to the global climate are likely to be a threat to smalltooth sawfish and the habitats they use. The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal and its impacts to coastal resources may be significant (Intergovernmental Panel on Climate Change 2007; Intergovernmental Panel on Climate Change 2014). Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, changes in the amount and timing of precipitation, and changes in air and water temperatures (NOAA 2012; USEPA 2012). The impacts to smalltooth sawfish cannot, for the most part, currently be predicted with any degree of certainty, but we can project some effects to the coastal habitats where they reside. Red mangroves and shallow, euryhaline waters will be directly impacted by climate change through sea level rise, which is expected to increase 0.45 to 0.75 m by 2100 (Intergovernmental Panel on Climate Change 2014). Sea level rise will impact mangrove resources, as sediment surface elevations for mangroves will not keep pace with conservative projected rates of elevation in sea level (Gilman et al. 2008). Sea level increases will also affect the amount of shallow water available for juvenile smalltooth sawfish nursery habitat, especially in areas where there is shoreline armoring (e.g., seawalls). Further, the changes in precipitation coupled with sea level rise may also alter salinities of coastal habitats, reducing the amount of available smalltooth sawfish nursery habitat.

4.1.6 Coral

There are 5 species of corals (elkhorn, staghorn, lobed star, boulder star, mountainous star,) in the action area that we believe may be adversely affected by activities covered under this Opinion that occur on shallow coral reefs (Figure 32) widely throughout wider-Caribbean, including south Florida, Puerto Rico, U.S. Virgin Islands, and the Gulf of Mexico (only star corals). Due to their broad distribution and sessile nature, these species may occur within the action area. Section 4.1.6.1 of this Opinion will address the general threats to all coral species. The rest of Section 4.1.6 of this Opinion will address the distribution, life history, population structure, abundance, population trends, and unique threats to each species of coral.

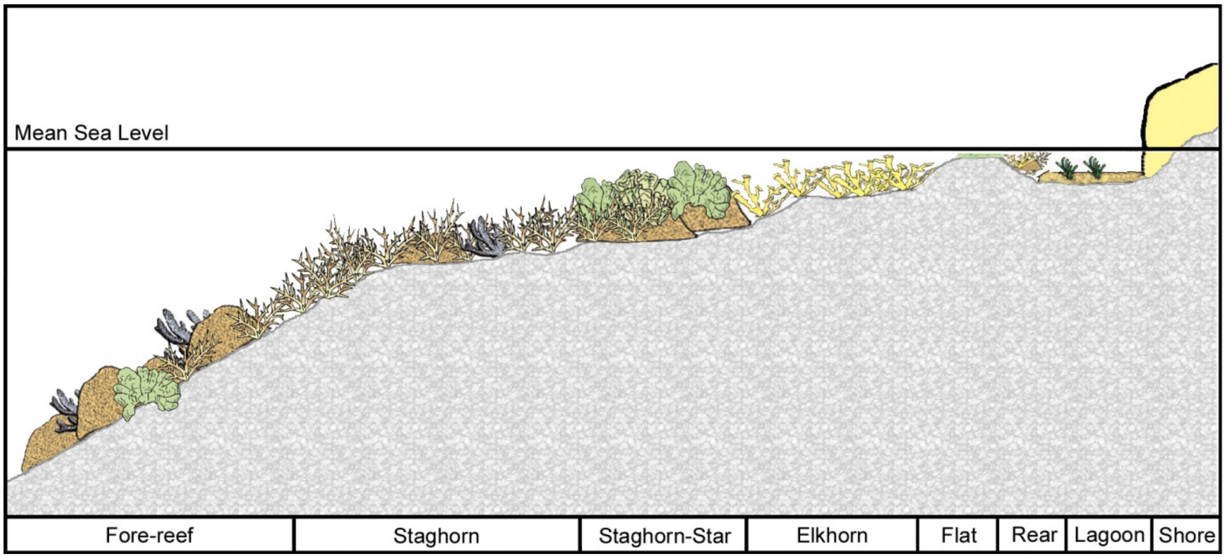

Figure 32. Reef zonation schematic example modified from several reef zonation-descriptive studies (Bak 1977; Goreau 1959).

4.1.6.1 General Threats Faced by All Coral Species

Corals face numerous natural and man-made threats that shape their status and affect their ability to recover. Many of the threats are either the same or similar in nature for all listed coral species, those identified in this section are discussed in a general sense for all corals. All threats are expected to increase in severity in the future. More detailed information on the threats to listed corals is found in the Final Listing Rule (79 FR 53851; Publication Date September 10, 2014). Threat information specific to a particular species is then discussed in the corresponding status sections where appropriate.

Several of the most important threats contributing to the extinction risk of corals are related to global climate change. The main concerns regarding impacts of global climate change on coral reefs generally, and on listed corals in particular, are the magnitude and the rapid pace of change in greenhouse gas concentrations (e.g., carbon dioxide [CO_2] and methane) and atmospheric warming since the Industrial Revolution in the mid-19th century. These changes are increasing the warming of the global climate system and altering the carbonate chemistry of the ocean (ocean acidification). Ocean acidification affects a number of biological processes in corals, including secretion of their skeletons.

Ocean Warming

Ocean warming is one of the most important threats posing extinction risks to the listed coral species, but individual susceptibility varies among species. The primary observable coral response to ocean warming is bleaching of adult coral colonies, wherein corals expel their symbiotic algae in response to stress. For many corals, an episodic increase of only 1°C–2°C above the normal local seasonal maximum ocean temperature can induce bleaching. Corals can withstand mild to moderate bleaching; however, severe, repeated, and/or prolonged bleaching can lead to colony death. Coral bleaching patterns are complex, with several species exhibiting seasonal cycles in symbiotic algae density. Thermal stress has led to bleaching and mass mortality in many coral species during the past 25 years.

In addition to coral bleaching, other effects of ocean warming can harm virtually every life-history stage in reef-building corals. Impaired fertilization, developmental abnormalities, mortality, impaired settlement success, and impaired calcification of early life phases have all been documented. Average seawater temperatures in reef-building coral habitat in the wider Caribbean have increased during the past few decades and are predicted to continue to rise between now and 2100. Further, the frequency of warm-season temperature extremes (warming events) in reef-building coral habitat has increased during the past 2 decades and is predicted to continue to increase between now and 2100.

Ocean Acidification

Ocean acidification is a result of global climate change caused by increased CO_2 in the atmosphere and dissolving into seawater. Reef-building corals produce skeletons made of the aragonite form of calcium carbonate. Ocean acidification reduces aragonite concentrations in seawater, making it more difficult for corals to build their skeletons. Ocean acidification has the potential to cause substantial reduction in coral calcification and reef cementation. Further, ocean acidification impacts adult growth rates and fecundity, fertilization, pelagic planula settlement, polyp development, and juvenile growth. Ocean acidification can lead to increased colony breakage, fragmentation, and mortality. Based on observations in areas with naturally low pH, the effects of increasing ocean acidification may also include reductions in coral size, cover, diversity, and structural complexity.

As CO_2 concentrations increase in the atmosphere, more CO_2 is absorbed by the oceans, causing lower pH and reduced availability of calcium carbonate. Because of the increase in CO_2 and other greenhouse gases in the atmosphere since the Industrial Revolution, ocean acidification has already occurred throughout the world's oceans, including in the Caribbean, and is predicted to increase considerably between now and 2100. Along with ocean warming and disease, we consider ocean acidification to be one of the most important threats posing extinction risks to coral species between now and the year 2100, although individual susceptibility varies among the listed corals.

Diseases

Disease adversely affects various coral life history events by, among other processes, causing adult mortality, reducing sexual and asexual reproductive success, and impairing colony growth. A diseased state results from a complex interplay of factors including the cause or agent (e.g., pathogen, environmental toxicant), the host, and the environment. All coral disease impacts are presumed to be attributable to infectious diseases or to poorly described genetic defects. Coral disease often produces acute tissue loss. Other forms of "disease" in the broader sense, such as temperature-caused bleaching, are discussed in other threat sections (e.g., ocean warming as a result of climate change).

Coral diseases are a common and significant threat affecting most or all coral species and regions to some degree, although the scientific understanding of individual disease causes in corals remains very poor. The incidence of coral disease appears to be expanding geographically, though the prevalence of disease is highly variable between sites and species. Increased prevalence and severity of diseases is correlated with increased water temperatures, which may correspond to increased virulence of pathogens, decreased resistance of hosts, or both. Moreover, the expanding coral disease threat may result from opportunistic pathogens that become damaging only in situations where the host integrity is compromised by physiological stress or immune suppression. Overall, there is mounting evidence that warming temperatures and coral bleaching responses are linked (albeit with mixed correlations) with increased coral disease prevalence and mortality.

Trophic Effects of Reef Fishing
Fishing, particularly overfishing, can have large-scale, long-term ecosystem-level effects that can change ecosystem structure from coral-dominated reefs to algal-dominated reefs ("phase shifts"). Even fishing pressure that does not rise to the level of overfishing potentially can alter trophic interactions that are important in structuring coral reef ecosystems. These trophic interactions include reducing population abundance of herbivorous fish species that control algal growth, limiting the size structure of fish populations, reducing species richness of herbivorous fish, and releasing corallivores from predator control.

In the Caribbean, parrotfishes can graze at rates of more than 150,000 bites per square meter per day (Carpenter 1986), and thereby remove up to 90-100% of the daily primary production (e.g., algae; Hatcher 1997). With substantial populations of herbivorous fishes, as long as the cover of living coral is high and resistant to mortality from environmental changes, it is very unlikely that the algae will take over and dominate the substrate. However, if herbivorous fish populations, particularly large-bodied parrotfish, are heavily fished and a major mortality of coral colonies occurs, then algae can grow rapidly and prevent the recovery of the coral population. The ecosystem can then collapse into an alternative stable state, a persistent phase shift in which algae replace corals as the dominant reef species. Although algae can have negative effects on adult coral colonies (e.g., overgrowth, bleaching from toxic compounds), the ecosystem-level effects of algae are primarily from inhibited coral recruitment. Filamentous algae can prevent the colonization of the substrate by planula larvae by creating sediment traps that obstruct access to a hard substrate for attachment. Additionally, macroalgae can block successful colonization of the bottom by corals because the macroalgae takes up the available space and causes shading, abrasion, chemical poisoning, and infection with bacterial disease. Trophic effects of fishing are a medium importance threat to the extinction risk for listed corals.

Sedimentation
Human activities in coastal and inland watersheds introduce sediment into the ocean by a variety of mechanisms including river discharge, surface runoff, groundwater seeps, and atmospheric deposition. Humans also introduce sewage into coastal waters through direct discharge, treatment plants, and septic leakage. Elevated sediment levels are generated by poor land use practices and coastal and nearshore construction.

The most common direct effect of sedimentation is sediment's landing on coral surfaces as it settles out from the water column. Corals with certain morphologies (e.g., mounding) can passively reject settling sediments. In addition, corals can actively remove sediment but at a significant energy cost. Corals with large calices (skeletal component that holds the polyp) tend to be better at actively rejecting sediment. Some coral species can tolerate complete burial for several days. Corals that cannot remove sediment will be smothered and die. Sediment can also cause sublethal effects such as reductions in tissue thickness, polyp swelling, zooxanthellae loss, and excess mucus production. In addition, suspended sediment can reduce the amount of light in the water column, making less energy available for coral photosynthesis and growth. Sedimentation also impedes fertilization of spawned gametes and reduces larval settlement and survival of recruits and juveniles.

Nutrient Enrichment

Elevated nutrient concentrations in seawater affect corals through 2 main mechanisms: direct impacts on coral physiology, and indirect effects through stimulation of other community components (e.g., macroalgal turfs and seaweeds, and filter feeders) that compete with corals for space on the reef. Increased nutrients can decrease calcification; however, nutrients may also enhance linear extension while reducing skeletal density. Either condition results in corals that are more prone to breakage or erosion, but individual species do have varying tolerances to increased nutrients. Anthropogenic nutrients mainly come from point-source discharges (such as rivers or sewage outfalls) and surface runoff from modified watersheds. Natural processes, such as *in situ* nitrogen fixation and delivery of nutrient-rich deep water by internal waves and upwelling, also bring nutrients to coral reefs.

4.1.6.2 Elkhorn coral (*Acropora palmata*)

Elkhorn coral was listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing its status from threatened to endangered (77 FR 73219). On September 10, 2014, NMFS determined that elkhorn coral should remain listed as threatened (79 FR 53851).

4.1.6.2.1 Species Description and Distribution

Elkhorn coral colonies have frond-like branches, which appear flattened to near round, and typically radiate out from a central trunk and angle upward. Branches are up to approximately 20 inches (50 cm) wide and range in thickness from about 1.5-2 inches (4 to 5 cm). Individual colonies can grow to at least 6.5 ft (2 m) in height and 13 ft (4 m) in diameter (*Acropora* Biological Review Team 2005). Colonies of elkhorn coral can grow in nearly single-species, dense stands and form an interlocking framework known as thickets.

Elkhorn coral is distributed throughout the western Atlantic Ocean, Caribbean Sea, and Gulf of Mexico. The northern extent of the range in the Atlantic is Broward County, Florida, where it is relatively rare (only a few known colonies), but fossil elkhorn coral reef framework extends into Palm Beach County, Florida. There are 2 known colonies of elkhorn coral, which were discovered in 2003 and 2005, at the Flower Garden Banks, which is located 100 miles (161 km) off the coast of Texas in the Gulf of Mexico (Zimmer et al. 2006). The species has been affected by extirpation from many localized areas throughout its range (Jackson et al. 2014).

Goreau (1959) described 10 habitat zones on a Jamaican fringing reef from inshore to the deep slope, finding elkhorn coral in 8 of the 10 zones. Elkhorn coral commonly grows in turbulent water on the fore-reef, reef crest, and shallow spur-and-groove zone (Cairns 1982; Miller et al. 2008; Rogers et al. 1982; Shinn 1963) in water ranging from approximately 3-15 ft (1-5 m) depth, and up to 40 ft (12m). Elkhorn coral often grows in thickets in fringing and barrier reefs

(Jaap 1984; Tomascik and Sander 1987; Wheaton and Jaap 1988). They have formed extensive barrier-reef structures in Belize (Cairns 1982), the greater and lesser Corn Islands, Nicaragua (Lighty et al. 1982), and Roatan, Honduras, and extensive fringing reef structures throughout much of the Caribbean (Adey 1978). Early studies termed the reef crest and adjacent seaward areas from the surface down to approximately 20 ft (5-6 m) depth the "palmata zone" because of the domination by the species (Goreau 1959; Shinn 1963). It also occasionally occurs in back-reef environments and in depths up to 98 ft (30 m).

4.1.6.2.2 Life History Information

Relative to other corals, elkhorn coral has a high growth rate allowing acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates, measured as skeletal extension of the end of branches, range from approximately 2-4 in (4-11 cm) per year (*Acropora* Biological Review Team 2005). However, growth rates in Curaçao have been reported to be slower today than they were several decades ago (Brainard et al. 2011). Annual growth has been found to be dependent on the size of the colony, and new recruits and juveniles typically grow at slower rates. Additionally, stressed colonies and fragments may also exhibit slower growth.

Elkhorn coral is a hermaphroditic broadcast spawning[54] species that reproduces sexually after the full moon of July, August, and/or September, depending on location and timing of the full moon (*Acropora* Biological Review Team 2005). Split spawning (spawning over a 2 month period) has been reported from the Florida Keys (Fogarty et al. 2012). The estimated size at sexual maturity is approximately 250 inches^2 (1,600 cm^2), and growing edges and encrusting base areas are not fertile (Soong and Lang 1992). Larger colonies have higher fecundity per unit area, as do the upper branch surfaces (Soong and Lang 1992). Although self-fertilization is possible, elkhorn coral is largely self-incompatible (Baums et al. 2005a; Fogarty et al. 2012).

Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies in the field. Rates of post-settlement mortality after 9 months are high based on settlement experiments (Szmant and Miller 2006). Laboratory studies have found that certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010). Laboratory experiments have shown that some individuals (i.e., genotypes) are sexually incompatible (Baums et al. 2013) and that the proportion of eggs fertilized increases with higher sperm concentration (Fogarty et al. 2012). Experiments using gametes collected in Florida and Belize showed that Florida corals had lower fertilization rates than those from Belize, possibly due to genotype incompatibilities (Fogarty et al. 2012).

Reproduction occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Bak and Criens 1982; Highsmith 1982; Lirman 2000; Miller et al. 2007; Wallace 1985). Storms can be a method of producing fragments to establish new colonies (Fong and Lirman 1995). Fragmentation is an important mode of reproduction in many reef-building corals, especially for branching species such as elkhorn coral (Highsmith 1982; Lirman 2000; Wallace 1985). However, in the Florida Keys where populations have declined, there have been reports of failure of asexual recruitment due to high fragment mortality after storms (Porter et al. 2012; Williams and Miller 2010; Williams et al. 2008).

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[54] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

The combination of relatively rapid skeletal growth rates and frequent asexual reproduction by fragmentation can enable effective competition within, and domination of, elkhorn coral in reef-high-energy environments such as reef crests. Rapid skeletal growth rates and frequent asexual reproduction by fragmentation facilitate potential recovery from disturbances when environmental conditions permit (Highsmith 1982; Lirman 2000). However, low sexual reproduction can lead to reduced genetic diversity and limits the capacity to repopulate sites distant from the parent.

4.1.6.2.3 Status and Population Dynamics

Information on elkhorn coral status and populations dynamics is spotty throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations were data exist.

There appear to be 2 discrete populations of elkhorn coral. Genetic samples from 11 locations throughout the Caribbean indicate that elkhorn coral populations in the eastern Caribbean (St. Vincent and the Grenadines, U.S. Virgin Islands, Curaçao, and Bonaire) have had little or no genetic exchange with populations in the western Atlantic and western Caribbean (Bahamas, Florida, Mexico, Panama, Navassa, and Puerto Rico) (Baums et al. 2005b). While Puerto Rico is more closely connected with the western Caribbean, it is an area of mixing with contributions from both regions (Baums et al. 2005b). Models suggest that the Mona Passage between the Dominican Republic and Puerto Rico acts as a filter for larval dispersal and gene flow between the eastern Caribbean and western Caribbean (Baums et al. 2006b).

The western Caribbean is characterized by genetically poor populations with lower densities (0.13 ± 0.08 colonies per m^2). The eastern Caribbean populations are characterized by denser (0.30 ± 0.21 colonies per m^2), genotypically richer stands (Baums et al. 2006a). Baums et al. (2006a) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment. They postulated these geographic differences in the contribution of reproductive modes to population structure may be related to habitat characteristics, possibly the amount of shelf area available.

Genotypic diversity is highly variable. At 2 sites in the Florida Keys, only one genotype per site was detected out of 20 colonies sampled at each site (Baums et al. 2005b). In contrast, all 15 colonies sampled in Navassa had unique genotypes (Baums et al. 2006a). Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotpyes out of 120 samples; Zubillaga et al. 2008) and in Bonaire and Curaçao (18 genotypes of 22 samples and 19 genotypes of 20 samples, respectively; Baums et al. 2006a). In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24% and 65% of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006a).

A genetic study found significant population structure in Puerto Rico locations (Mona Island, Desecheo Island, La Parguerain, La Parguera) both between reefs and between locations. The study suggests that there is a restriction of gene flow between some reefs in close proximity in the La Parguera reefs resulting in greater population structure (Garcia Reyes and Schizas 2010). A more recent study provided additional detail on the genetic structure of elkhorn coral in Puerto

Rico, as compared to Curaçao, the Bahamas, and Guadeloupe that found unique genotypes in 75% of the samples with high genetic diversity (Mège et al. 2015). The recent results support 2 separate populations of elkhorn coral in the eastern Caribbean and western Caribbean; however, there is less evidence for separation at Mona Passage, as found by Baums et al. Baums et al. (2006b).

Elkhorn coral was historically one of the dominant species on Caribbean reefs, forming large, monotypic thickets and giving rise to the "elkhorn" zone in classical descriptions of Caribbean reef morphology (Goreau 1959). However, mass mortality, apparently from white-band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011). This mass mortality occurred throughout the range of the species within all Caribbean countries and archipelagos, even on reefs and banks far from localized human influence (Aronson and Precht 2001; Wilkinson 2008). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events added to the decline of elkhorn coral (Brainard et al. 2011). In locations where historic quantitative data are available (Florida, Jamaica, U.S. Virgin Islands), there was a reduction of greater than 97% between the 1970s and early 2000s in elkhorn coral populations (*Acropora* Biological Review Team 2005).

Since the 2006 listing of elkhorn coral, continued population declines have occurred in some locations with certain populations of elkhorn coral decreasing up to an additional 50% or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008). In addition, Williams et al. (2008) reported asexual recruitment failure between 2004 and 2007 in the upper Florida Keys after a major hurricane season in 2005 where less than 5% of the fragments produced recruited into the population. In contrast, several studies describe elkhorn coral populations that are showing some signs of recovery or are stable including in the Turks and Caicos Islands (Schelten et al. 2006), U.S. Virgin Islands (Grober-Dunsmore et al. 2006; Mayor et al. 2006; Rogers and Muller 2012), Venezuela (Zubillaga et al. 2008), and Belize (Macintyre and Toscano 2007).

There is some density data available for elkhorn corals in Florida, Puerto Rico, the U.S. Virgin Islands, and Cuba. In Florida, elkhorn coral was detected at 0% to 78% of the sites surveyed between 1999 and 2017. Average density ranged from 0.001 to 0.12 colonies per m^2 (NOAA, unpublished data). Elkhorn coral was encountered less frequently during benthic surveys in the U.S. Virgin Islands from 2002 to 2017. It was observed at 0 to 7% of surveyed reefs, and average density ranged from 0.001 to 0.01 colonies per m^2 (NOAA, unpublished data). Maximum elkhorn coral density at ten sites in St. John, U.S. Virgin Islands was 0.18 colonies per m^2 (Muller et al. 2014). In Puerto Rico, average density ranged from 0.002 to 0.09 colonies per m^2 in surveys conducted between 2008 and 2018, and elkhorn coal was observed on 1% to 27% of surveyed sites (NOAA, unpublished data). Density estimates from sites in Cuba range from 0.14 colonies per m^2 (Alcolado et al. 2010) to 0.18 colonies per m^2 (González-Díaz et al. 2010).

Mayor et al. (2006) reported the abundance of elkhorn coral in Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands. They surveyed 617 sites from May to June 2004 and extrapolated density observed per habitat type to total available habitat. Within an area of 795 ha, they estimated 97,232–134,371 (95% confidence limits) elkhorn coral colonies with any dimension of connected live tissue greater than 1 m. Mean densities (colonies $\geq$ 1 m) were 0.019 colonies per m^2 in branching coral-dominated habitats and 0.013 colonies per m^2 in other hardbottom habitats.

Puerto Rico contains the greatest known extent of elkhorn coral in the U.S. Caribbean; however, the species is still rarely encountered. Between 2006 and 2007, a survey of 431 random points in habitat suitable for elkhorn coral in 6 marine protected areas in Puerto Rico revealed a variable density of 0-52 elkhorn coral colonies per 100 m^2, with average density of 0.03 colonies per m^2. Live elkhorn coral colonies were present at 31% of all points sampled, and total loss of elkhorn coral was evidenced in 14% of the random survey areas where only dead standing colonies were present (Schärer et al. 2009).

In stratified random surveys along the south, southeast, southwest, and west coasts of Puerto Rico designed to locate *Acropora* colonies, elkhorn coral was observed at 5 out of 301 stations with sightings outside of the survey area at an additional 2 stations (García Sais et al. 2013). Elkhorn coral colonies were absent from survey sites along the Southeast coast. Maximum density was 18 colonies per 15 m^2 (1.2 colonies per m^2), and maximum colony size was approximately 7.5 ft (2.3 m) in diameter (García Sais et al. 2013).

Demographic monitoring of elkhorn coral colonies in Florida has shown a decline over time. Upper Florida Keys colonies showed more than 50% loss of tissue as well as a decline in the number of colonies, and a decline in the dominance by large colonies between 2004 and 2010 (Vardi et al. 2012; Williams and Miller 2012). Elasticity analysis from a population model based on data from the Florida Keys has shown that the largest individuals have the greatest contribution to the rate of change in population size (Vardi et al. 2012). Between 2010 and 2013, elkhorn coral in the middle and lower Florida Keys had mixed trends. Population densities remained relatively stable at 2 sites and decreased at 2 sites by 21% and 28% (Lunz 2013). Following the 2014 and 2015 thermal stress events, monitored elkhorn coral colonies lost one-third of their live tissue (Williams et al. 2017).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 45% to 77% of elkhorn corals were impacted (NOAA 2018b). Survey data for impacts to elkhorn corals are not available for the U.S. Virgin Islands or Florida, though qualitative observations indicate that damage was also widespread but variable by site.

At 8 of 11 sites in St. John, U.S. Virgin Islands, colonies of elkhorn coral increased in abundance, between 2001 and 2003, particularly in the smallest size class, with the number of colonies in the largest size class decreasing (Grober-Dunsmore et al. 2006). Colonies of elkhorn coral monitored monthly between 2003 and 2009 in Haulover Bay on St. John, U.S. Virgin Islands suffered bleaching and mortality from disease but showed an increase in abundance and size at the end of the monitoring period (Rogers and Muller 2012). The overall density of elkhorn coral colonies around St. John did not significantly differ between 2004 and 2010 with 6 out of the 10 sites showing an increase in colony density. Size frequency distribution did not significantly change at 7 of the 10 sites, with 2 sites showing an increased abundance of large-sized (> 51 cm) colonies (Muller et al. 2014).

In Curaçao, elkhorn coral monitored between 2009 and 2011 decreased in abundance and increased in colony size, with stable tissue abundance following hurricane damage (Bright et al. 2013). The authors explained that the apparently conflicting trends of increasing colony size but similar tissue abundance likely resulted from the loss of small-sized colonies that skewed the distribution to larger size classes, rather than colony growth.

Simulation models using data from matrix models of elkhorn coral colonies from specific sites in Curaçao (2006-2011), the Florida Keys (2004-2011), Jamaica (2007-2010), Navassa (2006 and 2009), Puerto Rico (2007 and 2010), and the British Virgin Islands (2006 and 2007) indicate that most of these studied populations will continue to decline in size and extent by 2100 if environmental conditions remain unchanged (i.e., disturbance events such as hurrricanes do not increase; Vardi 2011). In contrast, the studied populations in Jamaica were projected to increase in abundance, and studied populations in Navassa were projected to remain stable. Studied populations in the British Virgin Islands were predicted to decrease slightly from their initial very low levels. Studied populations in Florida, Curaçao, and Puerto Rico were predicted to decline to zero by 2100. Because the study period did not include physical damage (storms), the population simulations in Jamaica, Navassa, and the British Virgin Islands may have contributed to the differing projected trends at sites in these locations.

A report on the status and trends of Caribbean corals over the last century indicates that cover of elkhorn coral has remained relatively stable at approximately 1% throughout the region since the large mortality events of the 1970s and 1980s. The report also indicates that the number of reefs with elkhorn coral present steadily declined from the 1980s to 2000-2004, then remained stable between 2000-2004 and 2005-2011. Elkhorn coral was present at about 20% of reefs surveyed in both the 5-year period of 2000-2004 and the 7-year period of 2005-2011. Elkhorn coral was dominant on approximately 5 to 10% of hundreds of reef sites surveyed throughout the Caribbean during the 4 periods of 1990-1994, 1995-1999, 2000-2004, and 2005-2011 (Jackson et al. 2014).

Overall, frequency of occurrence decreased from the 1980s to 2000, stabilizing in the first decade of 2000. There are locations such as the U.S. Virgin Islands where populations of elkhorn coral appear stable or possibly increasing in abundance and some such as the Florida Keys where population numbers are decreasing. In some cases when size class distribution is not reported, there is uncertainty of whether increases in abundance indicate growing populations or fragmentation of larger size classes into more small-sized colonies. From locations where size class distribution is reported, there is evidence of recruitment, but not the proportions of sexual versus asexual recruits. Events like hurricanes continue to heavily impact local populations and affect projections of persistence at local scales. We conclude there has been a significant decline of elkhorn coral throughout its range as evidenced by the decreased frequency of occurrence and that population abundance is likely to decrease in the future with increasing threats.

4.1.6.2.4 Threats

A summary of threats to all corals is provided in Section 4.1.6.1 of this Opinion (General Threats Faced by All Coral Species). Detailed information on the threats to elkhorn coral can be found in the Final Listing rule (79 FR 53851, Publication Date September 10, 2014); however, a brief summary is provided here. Elkhorn coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing, depensatory population effects from rapid, drastic declines and low sexual recruitment, and anthropogenic and natural abrasion and breakage.

Elkhorn coral is highly susceptible to disease as evidenced by the mass-mortality event in the 1970s and 1980s. White pox seems to be more common today than white band disease. The effects of disease are spatially and temporally (both seasonally and inter-annually) variable. Results from longer-term monitoring studies in the U.S. Virgin Islands and the Florida Keys indicate that disease can be a major cause of both partial and total colony mortality.

Elkhorn coral is highly susceptible to ocean warming. High water temperatures affect elkhorn coral through bleaching, lowered resistance to disease, and effects on reproduction. Temperature-induced bleaching and mortality following bleaching are temporally and spatially variable. Bleaching associated with the high temperatures in 2005 had a large impact on elkhorn coral with 40 to 50 % of bleached colonies suffering either partial or complete mortality in several locations. Algal symbionts did not shift in elkhorn coral after the 1998 bleaching event indicating the ability to adapt to rising temperatures may not occur through this mechanism. However, elkhorn coral showed evidence of resistance to bleaching from warmer temperatures in some portions of its range under some circumstances (Little Cayman). Through the effects on reproduction, high temperatures can potentially decrease larval supply and settlement success, decrease average larval dispersal distances, and cause earlier larval settlement affecting gene flow among populations.

Elkhorn coral is susceptible to acidification through reduced growth, calcification, and skeletal density. The effects of increased carbon dioxide combined with increased nutrients appear to be much worse than either stressor alone.

There are few studies of the effects of nutrients on elkhorn coral. Field experiments indicate that the mean net rate of uptake of nitrate by elkhorn coral exceeds that of ammonium by a factor of 2 and that elkhorn coral does not uptake nitrite (Bythell 1990). In Vega Baja, Puerto Rico, elkhorn coral mortality increased to 52% concurrent with pollution and sedimentation associated with raw sewage and beach nourishment, respectively, between December 2008 and June 2009 (Hernández-Delgado et al. 2011). Mortality presented as patchy necrosis-like and white pox-like conditions that impacted local reefs following anthropogenic disturbances and was higher inside the shallow platform (52-69%) and closer to the source of pollution (81-97%) compared to the outer reef (34 to 37 percent; Hernández-Delgado et al. 2011). Elkhorn coral is sensitive to nutrients as evidenced by increased mortality after exposure to raw sewage. Elkhorn coral is highly susceptible to nutrient enrichment. Elkhorn coral is also sensitive to sedimentation due to its poor capability of removing sediment and its high reliance on clear water for nutrition. Sedimentation can also cause tissue mortality.

Predators can have an impact on elkhorn coral both through tissue removal and the potential to spread disease. Predation pressure is spatially variable and almost non-existent in some locations. However, the effects of predation can become more severe if colonies decrease in abundance and density, as predators focus on the remaining living colonies.

4.1.6.2.5 Summary of Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events. Elkhorn coral is highly susceptible to a number of threats, and cumulative effects of multiple threats are likely to exacerbate vulnerability to extinction. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because elkhorn coral is limited to an area with high localized human impacts and predicted increasing threats. Elkhorn coral occurs in turbulent water on the back reef, fore reef, reef crest, and spur and groove zone in water ranging from 1 to 30 m in depth. This moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that will, on local and regional scales, experience highly variable thermal regimes and ocean chemistry at any given point in time. Elkhorn coral has low sexual recruitment rates, which exacerbates vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated. In contrast, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. We anticipate that the population abundance is likely to decrease in the future with increasing threats.

4.1.6.3 Staghorn coral (Acropora cervicornis)

Staghorn coral was listed as threatened under the ESA in 2006 (71 FR 26852, Publication Date May 9, 2006). In 2012, NMFS proposed changing its status from threatened to endangered (77 FR 73219, Publication Date December 7, 2012). On September 10, 2014, NMFS determined that staghorn coral should remain listed as threatened (79 FR 53851, Publication Date September 10, 2014).

4.1.6.3.1 Species Description and Distribution

Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges from 0.1-2 inches (0.25-5 cm; Lirman et al. 2010), and linear branch growth rates have been reported to range between 1.2-4.5 inches (3-11.5 cm) per year (*Acropora* Biological Review Team 2005). The species can exist as isolated branches, individual colonies up to about 5 ft (1.5 m) diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (*Acropora* Biological Review Team 2005).

Staghorn coral is distributed throughout the Caribbean Sea, in the southwestern Gulf of Mexico, and in the western Atlantic Ocean. The fossil record indicates that during the Holocene epoch, staghorn coral was present as far north as Palm Beach County in southeast Florida (Lighty et al. 1978), which is also the northern extent of its current distribution (Goldberg 1973).

Staghorn coral commonly occurs in water ranging from 16 to 65 ft (5 to 20 m) in depth, though it occurs in depths of 16-30 m at the northern extent of its range, and has been rarely found to 60 m in depth. Staghorn coral naturally occurs on spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hardbottom habitats (Cairns 1982; Davis 1982; Gilmore and Hall 1976; Goldberg 1973; Jaap 1984; Miller et al. 2008; Wheaton and Jaap 1988). Historically it grew in thickets in water ranging from approximately 16-65 ft (5-20 m) in depth; though it has rarely been found to approximately 195 ft (60 m; Davis 1982; Jaap 1984; Jaap et al. 1989; Schuhmacher and Zibrowius 1985; Wheaton and Jaap 1988). At the northern extent of its range, it grows in deeper water (~53-99 ft [16-30 m]; Goldberg 1973). Historically, staghorn coral was one of the primary constructors of mid-depth (approximately 33-50 ft [10-15 m]) reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatan peninsula (Adey 1978). In the Florida Keys, staghorn coral occurs in various habitats but is most prevalent on patch reefs as opposed to their former abundance in deeper fore-reef habitats (i.e., 16-65 ft; Miller et al. 2008). There is no evidence of range constriction, though loss of staghorn coral at the reef level has occurred (*Acropora* Biological Review Team 2005).

Precht and Aronson (2004) suggest that coincident with climate warming, staghorn coral only recently re-occupied its historic range after contracting to south of Miami, Florida, during the late Holocene. They based this idea on the presence of large thickets off Ft. Lauderdale, Florida, which were discovered in 1998 and had not been reported in the 1970s or 1980s (Precht and Aronson 2004). However, because the presence of sparse staghorn coral colonies in Palm Beach County, north of Ft. Lauderdale, was reported in the early 1970s (though no thicket formation was reported; Goldberg 1973), there is uncertainty associated with whether these thickets were present prior to their discovery or if they recently appeared coincident with warming. The proportion of reefs with staghorn coral present decreased dramatically after the Caribbean-wide mass mortality in the 1970s and 1980s, indicating the spatial structure of the species has been affected by extirpation from many localized areas throughout its range (Jackson et al. 2014).

4.1.6.3.2 Life History Information

Relative to other corals, staghorn coral has a high growth rate that have allowed acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates, measured as skeletal extension of the end of branches, range from approximately 2-4 in (4-11 cm) per year (*Acropora* Biological Review Team 2005). Annual linear extension has been found to be dependent on the size of the colony. New recruits and juveniles typically grow at slower rates. Stressed colonies and fragments may also exhibit slower growth.

Staghorn coral is a hermaphroditic broadcast spawning species[55]. The spawning season occurs several nights after the full moon in July, August, or September depending on location and

[55] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

timing of the full moon, and may be split over the course of more than one lunar cycle (Szmant 1986; Vargas-Angel et al. 2006). The estimated size at sexual maturity is approximately 6 in (17 cm) branch length, and large colonies produce proportionally more gametes than small colonies (Soong and Lang 1992). Basal and branch tip tissue is not fertile (Soong and Lang 1992). Sexual recruitment rates are low, and this species is generally not observed in coral settlement studies. Laboratory studies have found that the presence of certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010).

Reproduction occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Tunnicliffe 1981). The combination of branching morphology, asexual fragmentation, and fast growth rates relative to other corals, can lead to persistence of large areas dominated by staghorn coral. The combination of rapid skeletal growth rates and frequent asexual reproduction by fragmentation can enable effective competition and can facilitate potential recovery from disturbances when environmental conditions permit. However, low sexual reproduction can lead to reduced genetic diversity and limits the capacity to repopulate spatially dispersed sites.

4.1.6.3.3 Status and Population Dynamics

Information on staghorn coral status and populations dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from 9 regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 310 miles (500 km) apart are genetically different from each other with low gene flow across the greater Caribbean. Fine-scale genetic differences have been detected at reefs separated by as little as 1.25 miles (2 km), suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A. prolifera* with staghorn coral (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). Populations in Florida and Honduras are genetically distinct from each other and other populations in the U.S. Virgin Islands, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the U.S. Virgin Islands and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

Staghorn coral historically was one of the dominant species on most Caribbean reefs, forming large, single-species thickets and giving rise to the nominal distinct zone in classical descriptions of Caribbean reef morphology (Goreau 1959). Massive, Caribbean-wide mortality, apparently primarily from white band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events has added to the decline of staghorn coral (Brainard et al. 2011). In locations where quantitative data are available (Florida, Jamaica, U.S. Virgin Islands, Belize), there was a reduction of approximately 92 to greater than 97% between the 1970s and early 2000s (*Acropora* Biological Review Team 2005).

Since the 2006 listing of staghorn coral as threatened, continued population declines have occurred in some locations with certain populations of both listed *Acropora* species decreasing up to an additional 50% or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008). Some small pockets of remnant robust populations have been reported in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005; Riegl et al. 2009), and Dominican Republic (Lirman et al. 2010). Additionally, Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 miles (70.2 km) of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold water die-off of the 1970s.

Riegl et al. (2009) monitored staghorn coral in photo plots on the fringing reef near Roatan, Honduras from 1996 to 2005. Staghorn coral cover declined from 0.42% in 1996 to 0.14% in 1999 after the Caribbean bleaching event in 1998 and mortality from run-off associated with a Category 5 hurricane. Staghorn coral cover further declined to 0.09% in 2005. Staghorn coral colony frequency decreased 71% between 1997 and 1999. In sharp contrast, offshore bank reefs near Roatan had dense thickets of staghorn coral with 31% cover in photo-quadrats in 2005 and appeared to survive the 1998 bleaching event and hurricane, most likely due to bathymetric separation from land and greater flushing. Modeling showed that under undisturbed conditions, retention of the dense staghorn coral stands on the banks off Roatan is likely with a possible increased shift towards dominance by other coral species. However, the authors note that because their data and the literature seem to point to extrinsic factors as driving the decline of staghorn coral, it is unclear what the future may hold for this dense population (Riegl et al. 2009).

Other studies of population dynamics show mixed trends. While cover of staghorn coral increased from 0.6% in 1995 to 10.5% in 2004 (Idjadi et al. 2006) and 44% in 2005 on a Jamaican reef, it collapsed after the 2005 bleaching event and subsequent disease to less than 0.5% in 2006 (Quinn and Kojis 2008). A cold water die-off across the lower to upper Florida Keys in January 2010 resulted in the complete mortality of all staghorn coral colonies at 45 of the 74 reefs surveyed (61%) (Schopmeyer et al. 2012). Walker et al. (2012) report increasing size of 2 thickets (expansion of up to 7.5 times the original size of one of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50%, highlighting the dynamic nature of staghorn coral distribution via fragmentation and re-attachment.

A report on the status and trends of Caribbean corals over the last century indicates that the percentage of reefs with staghorn coral present has decreased over time. The frequency of reefs at which staghorn coral was described as the dominant coral has remained stable. The number of reefs with staghorn coral present declined during the 1980s from approximately 50 to 30% of reefs and remained relatively stable at 30% through the 1990s. The number of reefs with staghorn coral present decreased to approximately 20% in 2000-2004 and approximately 10% in 2005-2011 (Jackson et al. 2014).

There is some density data available for reefs in U.S. jurisdiction. In Florida, staghorn coral was detected at 3% to 15% of the sites surveyed between 1999 and 2017. Average density ranged from 0.001 to 0.17 colonies per m^2. Staghorn coral was encountered less frequently during benthic surveys in the U.S. Virgin Islands from 2002 to 2017. It was typically observed at < 3% of surveyed reefs with the highest frequency of observance at 18% in 2012. Density ranged from <0.001 to 0.07 colonies per m^2 (NOAA, unpublished data).

Benthic surveys between 2008 and 2018 in Puerto Rico detected an average density of 0.001 to 0.17 colonies per m^2, and colonies were observed at 4% to 25% of the reefs surveyed (NOAA, unpublished data). Staghorn coral was observed in 21 out of 301 stations between 2011 and 2013 in stratified random surveys designed to detect *Acropora* colonies along the south, southeast, southwest, and west coasts of Puerto Rico (García Sais et al. 2013). Staghorn coral was also observed at 16 sites outside of the surveyed area. The largest colony was 24 inches (60 cm) and density ranged from 1-10 colonies per 162 ft^2 ($15m^2$)(García Sais et al. 2013).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 38% to 54% of staghorn corals were impacted (NOAA 2018b). In a post-hurricane survey of 57 sites in Florida, all of the staghorn coral colonies encountered were damaged by the hurricane (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

Overall, populations appear to consist mostly of isolated colonies or small groups of colonies compared to the vast thickets once prominent throughout its range. Thickets are a prominent feature at only a few known locations. Across the Caribbean, frequency of occurrence has decreased since the 1980s. There are examples of increasing trends in some locations (Dry Tortugas and southeast Florida), but not over larger spatial scales or longer time frames. Population model projections from Honduras at one of the only known remaining thickets indicate the retention of this dense stand under undisturbed conditions. If refuge populations are able to persist, it is unclear whether they would be able to repopulate nearby reefs as observed sexual recruitment is low. Thus, we conclude that the species has undergone substantial population decline and decreases in the extent of occurrence throughout its range. We anticipate that population abundance is likely to decrease in the future with increasing threats.

4.1.6.3.4 Threats

A summary of threats to all corals is provided in Section 4.1.6.1 of this Opinion. Detailed information on the threats to staghorn coral can be found in the Final Listing rule (79 FR 53851; Publication Date September 10, 2014); however, a brief summary is provided here. Staghorn coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, as well as susceptible to trophic effects of fishing, depensatory population effects from rapid, drastic declines and low sexual recruitment, and anthropogenic and natural abrasion and breakage.

Staghorn coral is highly susceptible to disease as evidenced by the mass-mortality event in the 1970s and 1980s. Although disease is both spatially and temporally variable, about 5-6% of staghorn coral colonies appear to be affected by disease at any one time, though incidence of disease has been reported to range from 0-32% and up to 72% during an outbreak. There is indication that some colonies may be resistant to white band disease. Staghorn coral is also susceptible to several other diseases including one that causes rapid tissue loss from multiple lesions (e.g., Rapid Wasting Disease, White Patch Disease). Because few studies track diseased colonies over time, determining the present-day colony and population level effects of disease is difficult. One study that monitored individual colonies during an outbreak found that disease can be a major cause of both partial and total colony mortality (Williams and Miller 2005).

Staghorn coral is highly susceptible to bleaching in comparison to other coral species, and mortality after bleaching events is variable. Algal symbionts did not shift in staghorn coral after the 1998 bleaching event, indicating the ability of this species to acclimatize to rising temperatures may not occur through this mechanism. Data from Puerto Rico and Jamaica following the 2005 Caribbean bleaching event indicate that temperature anomalies can have a large impact on total and partial mortality and reproductive output.

Staghorn coral is highly susceptible to acidification through reduced growth, calcification, and skeletal density. The effects of increased carbon dioxide combined with increased nutrients appear to be synergistically worse and caused 100% mortality in some combination in one laboratory study.

Staghorn coral has high susceptibility to sedimentation through its sensitivity to turbidity (reduced light results in lower photosynthesis by symbiotic algae, so there is less food for the coral), and increased run-off from land clearing has resulted in mortality of this species through smothering. In addition, laboratory studies indicate the combination of sedimentation and nutrient enrichment appears to be synergistically worse.

Staghorn coral is also highly susceptible to elevated nutrients, which can cause decreased growth in staghorn coral. The combined effects of nutrients with other stressors such as elevated carbon dioxide and sedimentation appear to be worse than the effects of nutrients alone, and can cause colony mortality in some combinations.

Predators can have a negative impact on staghorn coral through both tissue removal and the spread of disease. Predation pressure appears spatially variable. Removal of tissue from growing branch tips of staghorn coral may negatively affect colony growth, but the impact is unknown as most studies do not report on the same colonies through time, inhibiting evaluation of the longer-term impact of these predators on individual colonies and populations.

4.1.6.3.5 Summary of Status

The species has undergone substantial population decline and decreases in the extent of occurrence throughout its range due mostly to disease. There is evidence of synergistic effects of threats for this species where the effects of increased nutrients are combined with acidification and sedimentation. Staghorn coral is highly susceptible to a number of threats, and cumulative effects of multiple threats are likely to exacerbate vulnerability to extinction. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because staghorn coral is limited to areas with high, localized human impacts and predicted increasing threats. Staghorn coral commonly occurs in water ranging from 5 to 20 m in depth, though it occurs in depths of 16-30 m at the northern extent of its range, and has been rarely found to 60 m in depth. It occurs in spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hardbottom habitats. This habitat heterogeneity moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef and hardbottom environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Staghorn coral has low sexual recruitment rates, which exacerbates vulnerability to extinction due to decreased ability to recover from mortality events when all colonies at a site are extirpated. In contrast, its fast growth rates and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events, thus moderating vulnerability to extinction. Its abundance and life history characteristics, combined with spatial variability in ocean warming and acidification across the species' range, moderate the species' vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we also anticipate that the population abundance is likely to decrease in the future with increasing threats.

4.1.6.4 Boulder star coral (*Orbicella franksi*)

On September 10, 2014, NMFS listed boulder star coral as threatened (79 FR 53851, Publication Date September 10, 2019).

Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the 3 species in the *Orbicella annularis* (star coral) complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species –*Montastraea annularis*– with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata*, and *Montastraea* (now *Orbicella*) *franksi*. The 3 species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowlton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into 3 species.

Some studies report on the star coral species complex rather than individual species because visual distinction can be difficult where colony structure cannot be discerned (e.g., small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. Information about boulder star coral published prior to 1994 will be attributed to the species complex, since it is dated prior to the split of *Orbicella annularis* into 3 separate species.

4.1.6.4.1 Species Description and Distribution

Boulder star coral is distinguished by large, unevenly arrayed polyps that give the colony its characteristic irregular surface. Colony form is variable, and the skeleton is dense with poorly developed annual bands. Colony diameter can reach up to 16 ft (5 m) with a height of up to 6.5 ft (2 m).

Boulder star coral is distributed in the western Atlantic Ocean and throughout the Caribbean Sea including in the Bahamas, Bermuda, and the Flower Garden Banks. Boulder star coral tends to have a deeper distribution than the other 2 species in the *Orbicella* species complex. It occupies most reef environments and has been reported from water depths ranging from approximately 16-165 ft (5-50 m), with the species complex reported to 250 ft (90 m). *Orbicella* species are a common, often dominant, component of Caribbean mesophotic reefs (e.g., >100 ft [30 m]), suggesting the potential for deep refugia for boulder star coral.

4.1.6.4.2 Life History Information

The star coral species complex has growth rates ranging from 0.02-0.5 in (0.06-1.2 cm) per year and averaging approximately 0.3 in (1 cm) linear growth per year. Boulder star coral is reported to be the slowest of the 3 species in the complex (Brainard et al. 2011). They grow more slowly in deeper water and in less clear water.

All 3 species of the star coral complex are hermaphroditic broadcast spawners[56], with spawning concentrated on 6-8 nights following the full moon in late August, September, or early October, depending on timing of the full moon and location. Boulder star coral spawning is reported to be about 1- 2 hours earlier than lobed star coral and mountainous star coral. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Fertilization success measured in the field was generally below 15% for all 3 species, as it was closely linked to the number of colonies concurrently spawning. In Puerto Rico, minimum size at reproduction for the star coral species complex was 13 $inches^2$ (83 cm^2).

Successful recruitment by the star coral species complex appears to always have been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of approximately 130 ft^2 (12 m^2) of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex. Of 351 colonies of boulder star coral tagged in Bocas del Toro, Panama, larger colonies were noted to spawn more frequently than smaller colonies between 2002 and 2009 (Levitan et al. 2011).

Of 351 boulder star coral colonies observed to spawn at a site off Bocas del Toro, Panama, 324 were unique genotypes. Over 90% of boulder star coral colonies on this reef were the product of sexual reproduction, and 19 genetic individuals had asexually propagated colonies made up of 2 to 4 spatially adjacent clones of each. Individuals within a genotype spawned more synchronously than individuals of different genotypes. Additionally, within 16 ft (5 m), colonies nearby spawned more synchronously than farther spaced colonies, regardless of genotype. At distances greater than 16 ft (5 m), spawning was random between colonies (Levitan et al. 2011).

[56] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

In addition to low recruitment rates, boulder star corals have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies.

4.1.6.4.3 Status and Population Dynamics

Information on boulder star coral status and population dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Reported density is variable by location and habitat and is reported to range from 0.002 to 10.5 colonies per ~100 ft^2 (10 m^2). Benthic surveys conducted in Florida between 1999 and 2017 recorded an average density of 0.01 to 0.36 colonies per m^2, and boulder star coral was observed at 5% to 45% of surveyed sites (NOAA, unpublished data). In Puerto Rico, boulder star coral was observed at 3% to 50% of sites, and average density ranged from 0.002 to 0.13 colonies per m^2 in surveys conducted between 2008 and 2018 (NOAA, unpublished data). In the U.S. Virgin Islands, boulder star coral was present at a density of 0.02 to 0.24 colonies per m^2 at 19% to 69% of sites surveyed between 1999 and 2018 (NOAA unpublished data). Limited surveys in the Flower Garden Banks reported a relatively stable density of 0.91 to 1.05 colonies per m^2 between 2010 and 2015, and boulder star coral was present at 90% to 100% of surveyed sites (NOAA, unpublished data). In a survey of 31 sites in Dominica between 1999 and 2002, boulder star coral was present in 7% of the sites at less than 1% cover (Steiner 2003). On remote reefs off southwest Cuba, colony density was 0.08 colonies per ~100 ft^2 (10 m^2) at 38 reef-crest sites and 1.05 colonies per ~100 ft^2 (10 m^2) at 30 reef-front sites (Alcolado et al. 2010). The number of boulder star coral colonies in Cuba with partial colony mortality were far more frequent than those with no mortality across all size classes, except for 1 (i.e., less than ~20 inches [50 cm]) that had similar frequency of colonies with and without partial mortality (Alcolado et al. 2010).

Abundance at some sites in Curaçao and Puerto Rico appeared to be stable over an 8-10 year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at 9 sites off Mona and Desecheo Islands, Puerto Rico. In 1998, 4% of all corals at 6 sites surveyed off Mona Island were boulder star coral colonies, and approximately 5% were boulder star corals in 2008; at Desecheo Island, about 2% of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

Recent events have greatly impacted boulder star coral populations in Florida and the U.S. Caribbean. An unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species, including boulder star coral. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 10-14% of boulder star corals were impacted (NOAA 2018b). In Florida, approximately 23% of boulder star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

In some locations, colony size has decreased over the past several decades. Bruckner conducted a survey of 185 sites (2010 and 2011) in 5 countries (The Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) and reported the size of boulder star coral and lobed star coral colonies as significantly smaller than mountainous star coral. The total mean partial mortality of boulder star coral was 25%. Overall, the total live area occupied by boulder star coral declined by a mean of 38%, and mean colony size declined from 210-131 inches^2 (1356 cm^2 to 845 cm^2). At the same time, there was a 137% increase in small tissue remnants, along with a decline in the proportion of large (1,500 to 30,000 cm^2), completely alive colonies. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012).

Overall, abundance of boulder star coral appears stable in some locations and has declined in others. Although boulder star coral remains common, the buffering capacity of its life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We anticipate that population abundance is likely to decrease in the future with increasing threats.

4.1.6.4.4 Threats

A summary of threats to all corals is provided in Section 4.1.6.1 of this. Detailed information on the threats to boulder star coral can be found in the Final Listing Rule (79 FR 53851; Publication Date September 10, 2014); however, a brief summary is provided here. Boulder star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

Available information indicates that boulder star coral is highly susceptible to warming temperatures with a reported 88-90% bleaching frequency. Reported bleaching-related mortality from one study is high at 75%. There is indication that new algal symbiotic species establishment occurs after bleaching in boulder star coral.

In a 2010 cold-water event that affected south Florida, boulder star coral ranked as the 14th most susceptible coral species out of the 25 most abundant coral species. Average partial mortality was 8% in surveys from Martin County to the lower Florida Keys after the 2010 cold-water event compared to 0.4% average mortality during summer surveys between 2005 and 2009.

Although there is no species-specific information on the susceptibility of boulder star coral to ocean acidification, genus information indicates that the species complex has reduced growth and fertilization success under acidic conditions. Thus, we conclude boulder star coral survival likely has high susceptibility to ocean acidification.

Boulder star coral is often reported as among the species with the highest disease prevalence. Although there are few quantitative studies of the effects of disease on boulder star coral, there is evidence that partial mortality can average about 25-30% and that disease can cause shifts to smaller size classes. Thus, we conclude that boulder star coral is highly susceptible to disease.

Genus information indicates sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance. Genus level information also indicates boulder star coral is likely susceptible to nutrient enrichment through reduced growth rates and lower recruitment. Additionally, nutrient enrichment has been shown to increase the severity of yellow band disease in boulder star coral. Thus, we conclude that boulder star coral survival is highly susceptible to sedimentation and nutrient enrichment.

4.1.6.4.5 Summary of Status

Boulder star coral has undergone declines most likely from disease and warming-induced bleaching. There is evidence of synergistic effects of threats for this species including increased disease severity with nutrient enrichment. Boulder star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate vulnerability to extinction. Despite declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes as has been observed in locations in its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because boulder star coral is limited to areas with high localized human impacts and predicted increasing threats. Its depth range of approximately 16-165 ft (5-50 m), possibly up to 295 ft (90 m), moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than those in which the species occurs. Boulder star coral occurs in most reef habitats, including both shallow and mesophotic reefs, which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable temperatures and ocean chemistry at any given point in time.

However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

4.1.6.5 Lobed star coral (Orbicella annularis)

On September 10, 2014, NMFS listed lobed star coral as threatened (79 FR 53851, Publication Date September 10, 2014).

4.1.6.5.1 Species Description and Distribution

Lobed star coral colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other 2 star coral species, margins on the sides of columns are typically dead. Live colony surfaces usually lack ridges or bumps.

Lobed star coral is common throughout the western Atlantic Ocean and greater Caribbean Sea including the Flower Garden Banks, but may be absent from Bermuda. Lobed star coral is reported from most reef environments in depths of approximately 1.5-66 ft (0.5-20 m). The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >100 ft [30 m]) reefs, suggesting the potential for deep refuge across a broader depth range, but lobed star coral is generally described with a shallower distribution.

Asexual fission and partial mortality can lead to multiple clones of the same colony. The percentage of unique individuals is variable by location and is reported to range between 18% and 86% (thus, 14-82% are clones). Colonies in areas with higher disturbance from hurricanes tend to have more clonality. Genetic data indicate that there is some population structure in the eastern, central, and western Caribbean with population connectivity within but not across areas. Although lobed star coral is still abundant, it may exhibit high clonality in some locations, meaning that there may be low genetic diversity.

4.1.6.5.2 Life History Information

The star coral species complex has growth rates ranging from 0.02-0.5 in (0.06-1.2 cm) per year and averaging approximately 0.3 in (1 cm) linear growth per year. The reported growth rate of lobed star coral is 0.4 to 1.2 cm per year (Cruz-Piñón et al. 2003; Tomascik 1990). They grow more slowly in deeper water and in less clear water.

All 3 species of the star coral complex are hermaphroditic broadcast spawners[57], with spawning concentrated on 6-8 nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Further, mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about 1-2 hours earlier. Fertilization success measured in the field was generally below 15% for all 3 species, as it is closely linked to the number of colonies concurrently spawning. Lobed star coral is reported to have slightly smaller egg size and potentially smaller size/age at first reproduction that the other 2 species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 12 inches^2 (83 cm^2).

[57] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

Successful recruitment by the star coral complex species has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 130 ft^2 (12 m^2) of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

In addition to low recruitment rates, lobed star corals have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the lobed star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies.

4.1.6.5.3 Status and Population Dynamics

Information on lobed star coral status and populations dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Lobed star coral has been described as common overall. Demographic data collected in Puerto Rico over 9 years before and after the 2005 bleaching event showed that population growth rates were stable in the pre-bleaching period (2001–2005) but declined one year after the bleaching event. Population growth rates declined even further 2 years after the bleaching event, but they returned and then stabilized at the lower rate the following year.

Colony density varies by habitat and location, and ranges from less than 0.1 to greater than 1 colony per approximately 100 ft^2 (10 m^2). Benthic surveys along the Florida Reef Tract between 1999 and 2017 recorded an average density of 0.01 to 0.09 colonies per m^2, and lobed star coral was observed at 4% to 16% of surveyed sites (NOAA, unpublished data). Average density of lobed star corals in Puerto Rico ranged from 0.01 to 0.08 colonies per m^2 in surveys conducted between 2008 and 2018 and was observed at 9% to 63% of surveyed sites (NOAA, unpublished data). In the U.S. Virgin Islands, average density ranged from 0.03 to 0.21 colonies per m^2 in benthic surveys conducted between 2002 and 2017, and lobed star coral was observed at 25% to 54% of surveyed sites (NOAA, unpublished data). In the Flower Garden Banks, limited surveys detected lobed star corals at none to 24% of surveyed sites, and density was recorded as 0.1 colonies per m^2 in 2010 and 0.01 colonies per m^2 in 2013 (NOAA, unpublished data). Off southwest Cuba on remote reefs, average lobed star coral density was 0.31 colonies per approximately 108 ft^2 (10 m^2) at 38 reef-crest sites and 1.58 colonies per approximately 108 ft^2 (10 m^2) at 30 reef-front sites. Colonies with partial mortality were far more frequent than those with no partial mortality, which only occurred in the size class less than 40 inches (100 cm) (Alcolado et al. 2010).

Recent events have greatly impacted coral populations in Florida and the U.S. Caribbean. An unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016). Lobed star coral was one of the species in surveys that showed the highest prevalence of disease, and populations were reduced to < 25% of the initial population size (Precht et al. 2016).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 43-44% of lobed star corals were impacted (NOAA 2018b). In Florida, approximately 80% of lobed star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

Population trends are available from a number of studies. In a study of sites inside and outside a marine protected area in Belize, lobed star coral cover declined significantly over a 10-year period (1998/99 to 2008/09) (Huntington et al. 2011). In a study of 10 sites inside and outside of a marine reserve in the Exuma Cays, Bahamas, cover of lobed star coral increased between 2004 and 2007 inside the protected area and decreased outside the protected area (Mumby and Harborne 2010). Between 1996 and 2006, lobed star coral declined in cover by 37% in permanent monitoring stations in the Florida Keys (Waddell and Clarke (editors) 2008). Cover of lobed star coral declined 71% in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Star corals are the 3rd most abundant coral by percent cover in permanent monitoring stations in the U.S. Virgin Islands. A decline of 60% was observed between 2001 and 2012 primarily due to bleaching in 2005. However, most of the mortality was partial mortality, and colony density in monitoring stations did not change (Smith 2013).

Bruckner and Hill (2009) did not note any extirpation of lobed star coral at 9 sites off Mona and Desecheo Islands, Puerto Rico, monitored between 1995 and 2008. However, mountainous star coral and lobed star coral sustained the largest losses with the number of colonies of lobed star coral decreasing by 19% and 20% at Mona and Desecheo Islands, respectively. In 1998, 8% of all corals at 6 sites surveyed off Mona Island were lobed star coral colonies, dipping to approximately 6% in 2008. At Desecheo Island, 14% of all coral colonies were lobed star coral in 2000 while 13% were in 2008 (Bruckner and Hill 2009).

In a survey of 185 sites in 5 countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) in 2010 and 2011, size of lobed star coral and boulder star coral colonies was significantly smaller than mountainous star coral. Total mean partial mortality of lobed star coral colonies at all sites was 40%. Overall, the total area occupied by live lobed star coral declined by a mean of 51%, and mean colony size declined from 299 $inches^2$ to 146 $inches^2$ (1927 cm^2 to 939 cm^2). There was a 211% increase in small tissue remnants less than 78 $inches^2$ (500 cm^2), while the proportion of completely live large (1.6-32 ft^2 [1,500- 30,000 cm^2]) colonies declined. Star coral colonies in Puerto Rico were much larger with large amounts of dead sections. In contrast, colonies in Bonaire were also large with greater amounts of live tissue. The presence of dead sections was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish algal lawns (Bruckner 2012).

Cover of lobed star coral at Yawzi Point, St. John, U.S. Virgin Islands declined from 41% in 1988 to approximately 12% by 2003 as a rapid decline began with the aftermath of Hurricane Hugo in 1989 (Edmunds and Elahi 2007). This decline continued between 1994 and 1999 during a time of 2 hurricanes (1995) and a year of unusually high sea temperature (1998), but percent cover remained statistically unchanged between 1999 and 2003. Colony abundances declined from 47 to 20 colonies per approximately 10 ft^2 (1 m^2) between 1988 and 2003, due mostly to the death and fission of medium-to-large colonies ($\geq$ 24 $inches^2$ [151 cm^2]). Meanwhile, the population size class structure shifted between 1988 and 2003 to a higher proportion of smaller colonies in 2003 (60% less than 7 $inches^2$ [50 cm^2] in 1988 versus 70% in 2003) and lower proportion of large colonies (6% greater than 39 $inches^2$ [250 cm^2] in 1988 versus 3% in 2003). The changes in population size structure indicated a population decline coincident with the period of apparent stable coral cover. Population modeling forecasted the 1988 size structure would not be reestablished by recruitment and a strong likelihood of extirpation of lobed star coral at this site within 50 years (Edmunds and Elahi 2007).

Lobed star coral colonies were monitored between 2001 and 2009 at Culebra Island, Puerto Rico. The population was in demographic equilibrium (high rates of survival and stasis) before the 2005 bleaching event, but it suffered a significant decline in growth rate (mortality and shrinkage) for 2 consecutive years after the bleaching event. Partial tissue mortality due to bleaching caused dramatic colony fragmentation that resulted in a population made up almost entirely of small colonies by 2007 (97% were less than 7 $inches^2$ [50 cm^2]). Three years after the bleaching event, the population stabilized at about half of the previous level, with fewer medium-to-large size colonies and more smaller colonies (Hernández-Delgado et al. 2011).

Lobed star coral was historically considered to be one of the most abundant species in the Caribbean (Weil and Knowlton 1994). Percent cover has declined by 37% to 90% over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. Although star coral remains common in occurrence, abundance has decreased in some areas by 19% to 57%, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, U.S. Virgin Islands, and St. Kitts and Nevis. At some reefs, a large proportion of the population is comprised of non-fertile or less-reproductive size classes. Several population projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. Although lobed star coral is still common throughout the Caribbean, substantial population decline has occurred. The buffering capacity of lobed star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. Population abundance is likely to decrease in the future with increasing threats.

4.1.6.5.4 Threats

A summary of threats to all corals is provided in Section 4.1.6.1 of this Opinion. Detailed information on the threats to lobed star coral can be found in the Final Listing Rule (79 FR 53851; Publication Date September 10, 2014); however, a brief summary is provided here. Lobed star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

Lobed star coral is highly susceptible to bleaching with 45-100% of colonies observed to bleach. Reported mortality from bleaching ranges from 2-71%. Recovery after bleaching is slow with paled colonies observed for up to a year. Reproductive failure can occur a year after bleaching, and reduced reproduction has been observed 2 years post-bleaching. There is indication that new algal symbiotic species establishment can occur prior to, during, and after bleaching events and results in bleaching resistance in individual colonies. Thus, lobed star coral is highly susceptible to ocean warming.

In a 2010 cold-water event that affected south Florida, mortality of lobed star coral was higher than any other coral species in surveys from Martin County to the lower Florida Keys. Average partial mortality was 56% during the cold-water event compared to 0.3% from 2005 to 2009. Surveys at a Florida Keys inshore patch reef, which experienced temperatures less than 18˚C for 11 days, revealed lobed star coral was one of the most susceptible coral species with all colonies experiencing total colony mortality.

Although there is no species-specific information on the susceptibility of lobed star coral to ocean acidification, genus information indicates the species complex has reduced growth and fertilization success under acidic conditions. Thus, we conclude lobed star coral likely has high susceptibility to ocean acidification.

Lobed star coral is highly susceptible to disease. Most studies report lobed star coral as among the species with the highest disease prevalence. Disease can cause extensive loss in coral cover, high levels of partial colony mortality, and changes in the relative proportions of smaller and larger colonies, particularly when outbreaks occur after bleaching events.

Lobed star coral has high susceptibility to sedimentation. Sedimentation can cause partial mortality and decreased coral cover of lobed star coral. In addition, genus information indicates sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance. Lobed star coral also has high susceptibility to nutrients. Elevated nutrients cause increased disease severity in lobed star coral. Genus-level information indicates elevated nutrients also cause reduced growth rates and lowered recruitment.

4.1.6.5.5 Summary of Status

Lobed star coral has undergone major declines mostly due to warming-induced bleaching and disease. Several population projections indicate population decline in the future is likely at specific sites and that local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. There is evidence of synergistic effects of threats for this species, including disease outbreaks following bleaching events and increased disease severity with nutrient enrichment. Lobed star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate vulnerability to extinction. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. However, the buffering capacity of these life history characteristics is expected to decrease as colonies shift to smaller size classes, as has been observed in locations in the species' range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because lobed star coral is limited to areas with high localized human impacts and predicted increasing threats. Star coral occurs in most reef habitats 0.5-20 m in depth which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience high temperature variation and ocean chemistry at any given point in time. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

4.1.6.6 Mountainous star coral (*Orbicella faveolata*)

On September 10, 2014, NMFS listed mountainous star coral as threatened (79 FR 53851, Publication Date September 10, 2014).

4.1.6.6.1 Species Description and Distribution

Mountainous star coral grows in heads or sheets, the surface of which may be smooth or have keels or bumps. The skeleton is much less dense than in the other 2 star coral species. Colony diameters can reach up to 33 ft (10 m) with heights of 13-16 ft (4-5 m).

Mountainous star coral occurs in the western Atlantic and throughout the Caribbean, including Bahamas, Flower Garden Banks, and the entire Caribbean coastline. There is conflicting information on whether or not it occurs in Bermuda. Mountainous star coral has been reported in most reef habitats and is often the most abundant coral at 33-66 ft (10-20 m) in fore-reef environments. The depth range of mountainous star coral has been reported as approximately 1.5-132 ft (0.5-40 m), though the species complex has been reported to depths of 295 ft (90 m), indicating mountainous star coral's depth distribution is likely deeper than 132 ft (40 m). Star coral species are a common, often dominant component of Caribbean mesophotic reefs (e.g., > 100 ft [30 m]), suggesting the potential for deep refugia for mountainous star coral.

4.1.6.6.2 Life History Information

The star coral species complex has growth rates ranging from 0.02-0.5 inch (0.06-1.2 cm) per year and averaging approximately 0.3 inch (1 cm) linear growth per year. Mountainous star coral's growth rate is intermediate between the other star coral complex species (Szmant et al. 1997). They grow more slowly in deeper water and in water that is less clear.

The star coral complex species are hermaphroditic broadcast spawners,[58] as spawning is concentrated on 6-8 nights following the full moon in late August, September, or early October, depending on location and timing of full moon. All 3 species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about 1-2 hours earlier. Fertilization success measured in the field was generally below 15% for all 3 species, as it is closely linked to the number of colonies concurrently spawning. In Puerto Rico, minimum size at reproduction for the star coral species complex was 12 inches^2 (83 cm^2).

Successful recruitment by the star coral species complex has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 130 ft^2 (12 m^2) of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

Life history characteristics of mountainous star coral is considered intermediate between lobed star coral and boulder star coral especially regarding growth rates, tissue regeneration, and egg size. Spatial distribution may affect fecundity on the reef, with deeper colonies of mountainous star coral being less fecund due to greater polyp spacing. Reported growth rates of mountainous star coral range between 0.12 and 0.64 in (0.3 and 1.6 cm) per year (Cruz-Piñón et al. 2003; Tomascik 1990; Villinski 2003; Waddell (editor) 2005). Graham and van Woesik (2013) report that 44% of small colonies of mountainous star coral in Puerto Morelos, Mexico that resulted from partial colony mortality produced eggs at sizes smaller than those typically characterized as being mature. The number of eggs produced per unit area of smaller fragments was significantly less than in larger size classes. Szmant and Miller (2006) reported low post-settlement survivorship for mountainous star coral transplanted to the field with only 3-15% remaining alive after 30 days. Post-settlement survivorship was much lower than the 29% observed for elkhorn coral after 7 months (Szmant and Miller 2006).

Mountainous star coral has slow growth rates, late reproductive maturity, and low recruitment rates. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the star coral species complex in the past (Bruckner 2012). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, we conclude that the buffering capacity of this life history strategy has been reduced by recent population declines and partial mortality, particularly in large colonies.

4.1.6.6.3 Status and Population Dynamics

Information on mountainous star coral status and populations dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring has not

[58] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

been conducted. Thus, the status and populations dynamics must be inferred from the few locations where data exist.

Information regarding population structure is limited. Observations of mountainous star coral from 182 sample sites in the upper and lower Florida Keys and Mexico showed 3 well-defined populations based on 5 genetic markers, but the populations were not stratified by geography, indicating they were shared among the 3 regions (Baums et al. 2010). Of 10 mountainous star coral colonies observed to spawn at a site off Bocas del Toro, Panama, there were only 3 genotypes (Levitan et al. 2011) potentially indicating 30% clonality.

Benthic surveys along the Florida Reef Tract between 1999 and 2017 have shown a decrease of mountainous star coral (NOAA, unpublished data). In 1999, mountainous star coral was present at 62% of surveyed sites and had an average density of 0.62 colonies per m^2. Presence and density decreased substantially after 2005, and in 2017, mountainous star coral was present at 30% of sites and had an average density of 0.09 colonies per m^2.

Benthic survey data for the U.S. Caribbean show less variability in the density of mountainous star coral. In Puerto Rico, average density was between 0.1 and 0.2 colonies per m^2 between 2008 and 2016 (NOAA, unpublished data). In 2018, average density was recorded as 0.01 colonies per m^2, the lowest recorded for all survey years. In the U.S. Virgin Islands, density ranged from 0.01 to 0.2 colonies per m^2 between 2002 and 2017 with no obvious trends among years.

Recent events have greatly impacted coral populations in Florida and the U.S. Caribbean. An unprecedented, multi-year disease event, which began in 2014, swept through Florida and caused massive mortality from St. Lucie Inlet in Martin County to Looe Key in the lower Florida Keys. The effects of this widespread disease have been severe, causing mortality of millions of coral colonies across several species, including mountainous star coral. At study sites in southeast Florida, prevalence of disease was recorded at 67% of all coral colonies and 81% of colonies of those species susceptible to the disease (Precht et al. 2016).

Hurricanes Irma and Maria caused substantial damage in Florida, Puerto Rico, and the U.S. Virgin Islands in 2017. Hurricane impacts included large, overturned and dislodged coral heads and extensive burial and breakage. At 153 survey locations in Puerto Rico, approximately 12-14% of mountainous star corals were impacted (NOAA 2018b). In Florida, approximately 24% of mountainous star corals surveyed at 57 sites were impacted (Florida Fish and Wildlife Conservation Commission, unpublished data). Survey data are not available for the U.S. Virgin Islands, though qualitative observations indicate that damage was also widespread but variable by site.

In the Flower Garden Banks, limited benthic surveys show density of mountainous star coral remained relatively stable between 2010 and 2015 (NOAA, unpublished data). Average density was recorded as 0.09 colonies per m^2 in 2010, 0.19 colonies per m^2 in 2013, and 0.21 colonies per m^2 in 2015. These may represent an increasing trend as the presence of mountainous star coral also increased during this same period. It was present at 35% of sites in 2010 and increased to 68% of sites in 2013 and 77% of sites in 2015.

Limited data are available for other areas of the Caribbean. On remote reefs off southwest Cuba, average density of mountainous star coral was 0.12 colonies per 108 ft^2 (10 m^2) at 38 reef-crest sites and 1.26 colonies per 108 ft^2 (10 m^2) at 30 reef-front sites (Alcolado et al. 2010). In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80% of the sites at 1-10% cover (Steiner 2003).

Population trend data exists for several locations. At 9 sites off Mona and Desecheo Islands, Puerto Rico, no species extirpations were noted at any site over 10 years of monitoring between 1998 and 2008 (Bruckner and Hill 2009). Both mountainous star coral and lobed star coral sustained large losses during the period. The number of colonies of mountainous star coral decreased by 36% and 48% at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27% of all corals at 6 sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11% in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12% of all coral colonies were mountainous star coral in 2000, compared to 7% in 2008.

In a survey of 185 sites in 5 countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) between 2010 and 2011, size of mountainous star coral colonies was significantly greater than boulder star coral and lobed star coral. The total mean partial mortality of mountainous star coral at all sites was 38%. The total live area occupied by mountainous star coral declined by a mean of 65%, and mean colony size declined from 43 ft^2 to 15 ft^2 (4005 cm^2 to 1413 cm^2). At the same time, there was a 168% increase in small tissue remnants less than 5 ft^2 (500 cm^2), while the proportion of completely live large (1.6 ft^2 to 32 ft^2 [1,500- 30,000 cm^2]) colonies decreased. Mountainous star coral colonies in Puerto Rico were much larger and sustained higher levels of mortality compared to the other 4 countries. Colonies in Bonaire were also large, but they experienced much lower levels of mortality. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012).

Overall, it appears that populations of mountainous star coral have been decreasing. Population decline has occurred over the past few decades with a 65% loss in mountainous star coral cover across 5 countries. Losses of mountainous star coral from Mona and Desecheo Islands, Puerto Rico include a 36-48% reduction in abundance and a decrease of 42-59% in its relative abundance (i.e., proportion relative to all coral colonies). High partial mortality of colonies has led to smaller colony sizes and a decrease of larger colonies in some locations such as The Bahamas, Bonaire, Puerto Rico, Cayman Islands, and St. Kitts and Nevis. We conclude that mountainous star coral has declined and that the buffering capacity of mountainous star coral's life history strategy, which has allowed it to remain abundant, has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

4.1.6.6.4 Threats

A summary of threats to all corals is provided in Section 4.1.6.1 of this Opinion. Detailed information on the threats to mountainous star coral can be found in the Final Listing Rule (79

FR 53851; Publication Date September 10, 2014); however, a brief summary is provided here. Mountainous star coral is highly susceptible to ocean warming, disease, ocean acidification, sedimentation, and nutrients, and susceptible to trophic effects of fishing.

Mountainous star coral is highly susceptible to elevated temperatures. In lab experiments, elevated temperatures resulted in misshapen embryos and differential gene expression in larvae that could indicate negative effects on larval development and survival. Bleaching susceptibility is generally high; 37-100% of mountainous star coral colonies have reported to bleach during several bleaching events. Chronic local stressors can exacerbate the effects of warming temperatures, which can result in slower recovery from bleaching, reduced calcification, and slower growth rates for several years following bleaching. Additionally, disease outbreaks affecting mountainous star coral have been linked to elevated temperature as they have occurred after bleaching events. We conclude that mountainous star coral is highly susceptible to elevated temperature.

Surveys at an inshore patch reef in the Florida Keys that experienced temperatures less than 18˚C for 11 days revealed species-specific cold-water susceptibility and low survivorship. Mountainous star coral was one of the more susceptible species with 90% of colonies experiencing total colony mortality, including some colonies estimated to be more than 200 years old (Kemp et al. 2011). In surveys from Martin County to the lower Florida Keys, mountainous star coral was the second most susceptible coral species, experiencing an average of 37% partial mortality (Lirman et al. 2011).

Mountainous star coral is highly susceptible to ocean acidification. Laboratory studies indicate that ocean acidification affects that mountainous star coral both through reduced fertilization of gametes and reduced growth of colonies (Carricart-Ganivet et al. 2012).

Mountainous star coral is often among the coral species with the highest disease prevalence and tissue loss. Outbreaks have been reported to affect 10-19% of mountainous star coral colonies, and yellow band disease and white plague have the greatest effect. Disease often affects larger colonies, and reported tissue loss due to disease ranges from 5-90%. Additionally, yellow band disease results in lower fecundity in diseased and recovered colonies of mountainous star coral. Therefore, we anticipate that mountainous star coral is highly susceptible to disease.

Sedimentation can cause partial mortality of mountainous star coral, and genus-level information indicates that sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance. Therefore, we anticipate that mountainous star coral is highly susceptible to sedimentation.

Although there is no species-specific information, the star coral species complex is susceptible to nutrient enrichment through reduced growth rates, lowered recruitment, and increased disease severity. Therefore, based on genus-level information, we anticipate that mountainous star coral is likely highly susceptible to nutrient enrichment.

4.1.6.6.5 Summary of Status

Mountainous star coral has undergone major declines mostly due to warming-induced bleaching and disease. There is evidence of synergistic effects of threats for this species including disease outbreaks following bleaching events and reduced thermal tolerance due to chronic local stressors stemming from land-based sources of pollution. Mountainous star coral is highly susceptible to a number of threats, and cumulative effects of multiple threats have likely contributed to its decline and exacerbate its vulnerability to extinction. Despite high declines, the species is still common and remains one of the most abundant species on Caribbean reefs. Its life history characteristics of large colony size and long life span have enabled it to remain relatively persistent despite slow growth and low recruitment rates, thus moderating vulnerability to extinction. The buffering capacity of these life history characteristics, however, is expected to decrease as colonies shift to smaller size classes as has been observed in locations in its range. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because mountainous star coral is limited to an area with high, localized human impacts and predicted increasing threats. Its depth range of 0.5 m to at least 40 m, possibly up to 90 m, moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters, and acidification is generally predicted to accelerate most in waters that are deeper and cooler than those in which the species occurs. Mountainous star coral occurs in most reef habitats, including both shallow and mesophotic reefs, which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable temperatures and ocean chemistry at any given point in time. Its abundance, life history characteristics, and depth distribution, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time. However, we anticipate that the population abundance is likely to decrease in the future with increasing threats.

4.1.7 Johnson's Seagrass

Johnson's seagrass (*Halophila johnsonii*) was identified as a species by Eiseman and McMillan in 1980 (Eiseman and McMillan 1980). Prior, *H. johnsonii* was referred to either as *H. decipiens* or *H. baillonis*, but based on morphological, anatomical, and phylogenetic information, it is most closely related to, and most closely resembles, *H. ovalis*, an Indo-Pacific, dioecious species (McMillan and Williams 1980; Posluszny and Tomlinson 1991; Waycott et al. 2002). NMFS listed Johnson's seagrass as threatened under the ESA on September 14, 1998 (63 FR 49035, 63 FR 49035 September 14, 1998).

4.1.7.1 Species Description and Distribution

Johnson's seagrass is a short-statured, shallow-rooted, seagrass species found in the intracoastal waters of southeastern Florida. It is characterized by pairs of linearly shaped leaves, each with a petiole (stalk) formed on the node (portion of the stem from which leaves grow) of a horizontally creeping rhizome (stem). Unbranched roots anchor the rhizome to the substrate at or just below the sediment surface. The leaves have smooth margins and are generally 2-5 cm in length (including the petioles). The distance between leaf pairs (internodes) rarely exceed 3-5 cm.

Johnson's seagrass has a narrow geographical range and has only been documented along approximately 200 km of coastline in southeastern Florida. *H. johnsonii* occurs just north of Sebastian Inlet (25.7478ºN, -80.128ºW) south to Virginia Key (25.7478°N, --80.1441°W) (Figure 33). This apparent endemism (uniqueness to a particular area) suggests that Johnson's seagrass has the most limited geographic distribution of any seagrass species in the world. Since the listing in 1998, there have been no observed reductions in the species' geographic extent. However, the St. Johns River Water Management District (SJRWMD) observed *H. johnsonii* 21 km north of Sebastian Inlet on the western shore of the Indian River Lagoon; a discovery that slightly extended the species' previously known range limit (Virnstein and Hall 2009).

Information on the species' distribution and results of limited experimental work suggest that Johnson's seagrass has a wider tolerance range for salinity, temperature, and optical water quality conditions than other species such as paddle grass, *Halophila decipiens* (Dawes et al. 1989; Durako et al. 2003; Gallegos and Kenworthy 1996; Kahn et al. 2013; Kenworthy and Fonseca 1996; Kenworthy and Haunert (editors) 1991; Torquemada et al. 2005)). It is an opportunistic plant that occurs in a patchy, disjunctive distribution from the intertidal zone to depths of approximately 2-3 m. Johnson's seagrass has been observed near the mouths of freshwater discharge canals (Gallegos and Kenworthy 1996), in deeper turbid waters of the interior portion of the Indian River Lagoon (Kenworthy 2000; Virnstein and Morris 2007a), and in clear water associated with the high energy environments and flood deltas inside ocean inlets (Heidelbaugh et al. 2000; Kenworthy 1993; Kenworthy 1997b; Virnstein and Morris 2007a; Virnstein et al. 1997). It can colonize and persist in high-tidal energy environments and has been observed where tidal velocities approach the threshold of motion for unconsolidated sediments (35-40 cm s^{-1}). The persistent presence of high-density, elevated patches of Johnson's seagrass on flood tidal deltas near inlets suggests that it is capable of sediment stabilization. Intertidal populations of Johnson's seagrass may be completely exposed at low tides, suggesting high tolerance to desiccation and wide temperature tolerance (Kahn and Durako 2009).

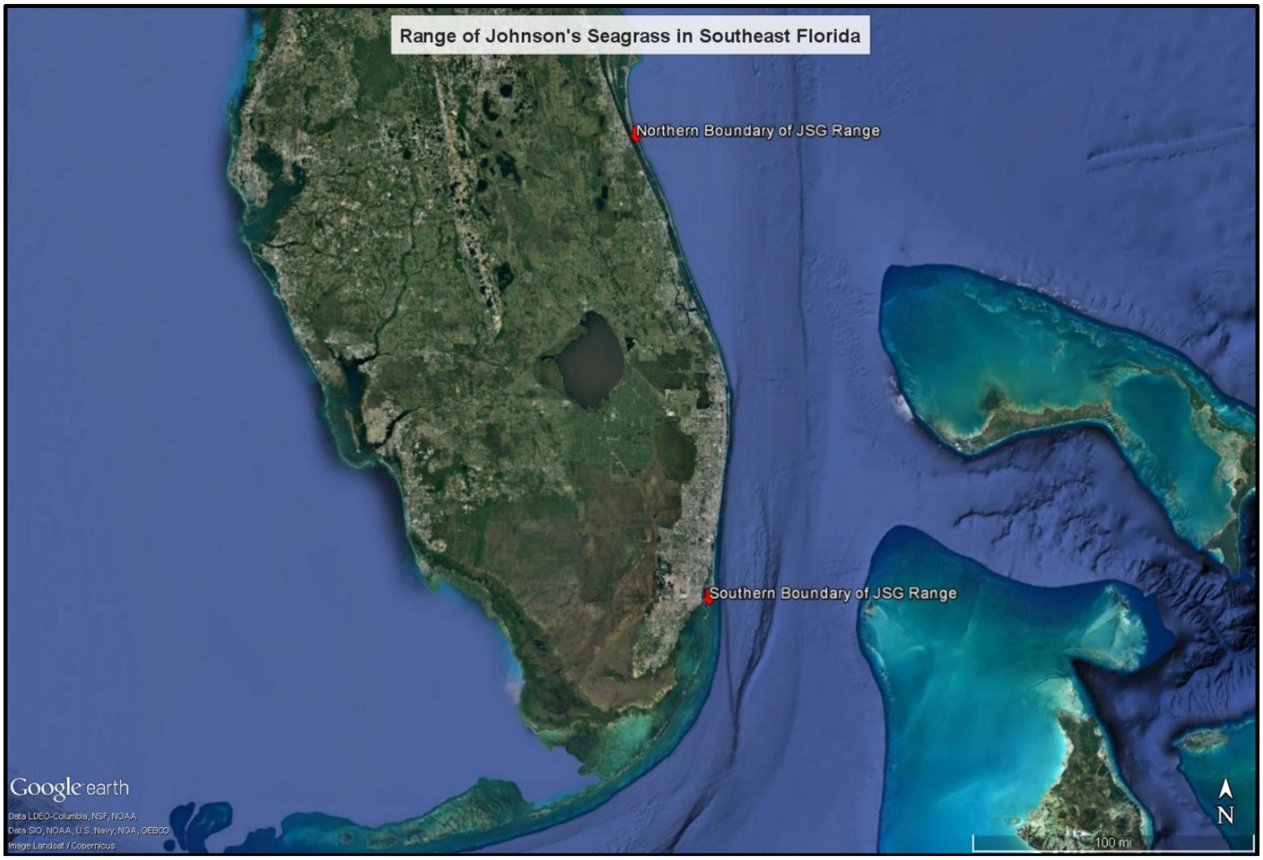

Figure 33. Range of Johnson's seagrass

4.1.7.2 Life History Information

Johnson's seagrass is a perennial species (meaning it lasts for greater than 2 growing seasons), showing no consistent seasonal or year-to-year pattern based on transect surveys. Johnson's seagrass reproduction is believed to be entirely asexual and dispersal is by vegetative fragmentation. Female flowers have been found; however, dedicated surveys in the Indian River Lagoon have not discovered male flowers, fertilized ovaries, fruits, or seeds (Hammerstom and Kenworthy 2003; Jewett-Smith et al. 1997; NMFS 2007a). Searches throughout the entire range of Johnson's seagrass have produced the same results, suggesting either that the species does not reproduce sexually or that the male flowers are difficult to observe or describe, as noted for other Halophila species (Kenworthy 1997b). Surveys to date indicate that the incidence of female flowers appears to be much higher near the inlets leading to the Atlantic Ocean.

Johnson's seagrass spreads rapidly, growing horizontally from dense apical meristems with leaf pairs having short life spans (Kenworthy 1997a). Kenworthy (Kenworthy 1997a)suggested that the observed horizontal spreading, rapid growth patterns, and high biomass turnover could explain the dynamic patches observed in distribution studies of this species. While patches may colonize quickly, they may also disappear rapidly. Sometimes they will disappear for several years and then re-establish, a process referred to as "pulsating patches" (Heidelbaugh et al. 2000; Virnstein and Hall 2009; Virnstein and Morris 2007a). Mortality, or the disappearance of patches, can be caused by a number of processes, including burial from bioturbation and sediment deposition (Heidelbaugh et al. 2000), erosion, herbivory, desiccation, and turbidity. In the absence of sexual reproduction, one possible explanation for the pulsating patches is dispersal and re-establishment of vegetative fragments, a process that commonly occurs in aquatic plants and has been demonstrated in other seagrasses (di Carlo et al. 2005; Philbrick and Les 1996), and was also confirmed by experimental mesocosm[59] studies with Johnson's seagrass (Hall et al. 2006).

4.1.7.3 Status and Population Dynamics

Throughout its range, Johnson's seagrass occurs in dynamic and disjunctive patches. Observations by researchers have suggested that Johnson's seagrass exploits unstable environments or newly-created unvegetated patches by exhibiting fast growth and support for all local ramets. It may quickly recruit to locally uninhabited patches through prolific lateral branching and fast horizontal growth. While these attributes may allow it to compete effectively in periodically disturbed areas, if the distribution of this species becomes limited to stable areas it may eventually be outcompeted by more stable-selected plants represented by the larger-bodied seagrasses (Durako et al. 2003). In addition, the physiological attributes of Johnson's seagrass may limit growth (i.e., spreading) over large areas of substrate if the substrate is somehow altered (e.g., dredged to a depth that would preclude future recruitment of Johnson's seagrass); therefore, its ability to recover from widespread habitat loss may be limited. The clonal and reproductive growth characteristics of Johnson's seagrass result in its distribution being patchy, non-contiguous, and temporally fluctuating. These attributes suggest that colonization between broadly disjunctive areas is likely difficult and that the species is vulnerable to becoming endangered if it is removed from large areas within its range by natural or anthropogenic means.

Two survey programs have monitored the presence and abundance of Johnson's seagrass within its range. One program, conducted by the St. Johns River Water Management District since 1994, continues to survey the northern section of the species' geographic range between Sebastian Inlet and Jupiter Inlet (Virnstein and Hall 2009; Virnstein and Morris 2007b). The second survey, initiated in 2006, monitored the southern range of the species between Jupiter Inlet and Virginia Key in Biscayne Bay annually through 2012 (Kunzelman 2007). Since the last status review, there has not been any reported reduction in the geographic range of the species but rather a slight increase in the known northern range has been observed (Virnstein and Hall 2009).

Based on the results of the southern transect sampling, it appears there is a relatively continuous, although patchy, distribution of the species from Jupiter Inlet to Virginia Key (NMFS 2007a). The largest reported contiguous patch of Johnson's seagrass in the southern range was observed in Lake Worth Lagoon and was estimated to be 30 acres (Kenworthy 1997b). The presence of Johnson's seagrass in northern Biscayne Bay (north of Virginia Key) is well documented. There have been no reports of this species further south of the currently known southern distribution. Findings from the southern transect sampling (summer 2006 and winter 2007) show little difference in the species' frequency or abundance between the summer and winter sampling period. The lower frequencies of Johnson's seagrass occurred at those sites where larger-bodied

[59] A mesocosm is an experimental tool that brings a small part of the natural environment under controlled conditions.

seagrasses (e.g., turtle grass [*Thalassia testudinum*], and manatee grass [*Syringodium filiforme*]) were more abundant (NMFS 2007a). The southern range transect data support some of the conclusions drawn from previous studies and other surveys. This is a rare species; however, it can be abundant where it does occur. Based on the results of the southern transect sampling, it appears that, although it is disjunctively distributed and patchy, there is some continuity in the southern distribution, at least during periods of relatively good environmental conditions and no significant large-scale disturbances (NMFS 2007a).

Within the Indian River Lagoon in the northern section of the species' range, Johnson's seagrass was found associated with other seagrass species or growing alone in the intertidal and, more commonly, at the deep edge of some transects in water depths down to 180 cm (L. Morris, St. Johns River Water Management District, unpublished data). Fixed-site transect surveys, conducted between 1995 and 2017, indicated variable occurrence of Johnson's seagrass through time but a clear decline since 2010 (Figure 34, L. Morris, St. Johns River Water Management District, unpublished data). The decrease in occurrence also corresponded with a decline in percent coverage as well (Figure 35, L. Morris, St. Johns River Water Management District, unpublished data). Depth of occurrence within these surveys ranged from 0.03 to 2.5 m. Where the species occurs, its distribution is patchy, both spatially and temporally. The decline in Johnson's seagrass began just prior to several years of poor water quality involving a persistent drought 2009-2011, a phytoplankton "superbloom" in 2011, and a brown tide event in 2014. The "superbloom" exceeded any past bloom event in both geographic scale, bloom intensity, and duration, creating a decline in water clarity and a significant seagrass die-off (Consortium 2015). The persistent poor water quality has affected all seagrass species in the Indian River Lagoon and recovery of seagrasses will depend on improved water quality. A consortium of environmental agencies developed a plan to investigate the "superbloom" in an effort resolve lingering water quality issues in the Indian River Lagoon (Consortium 2015).

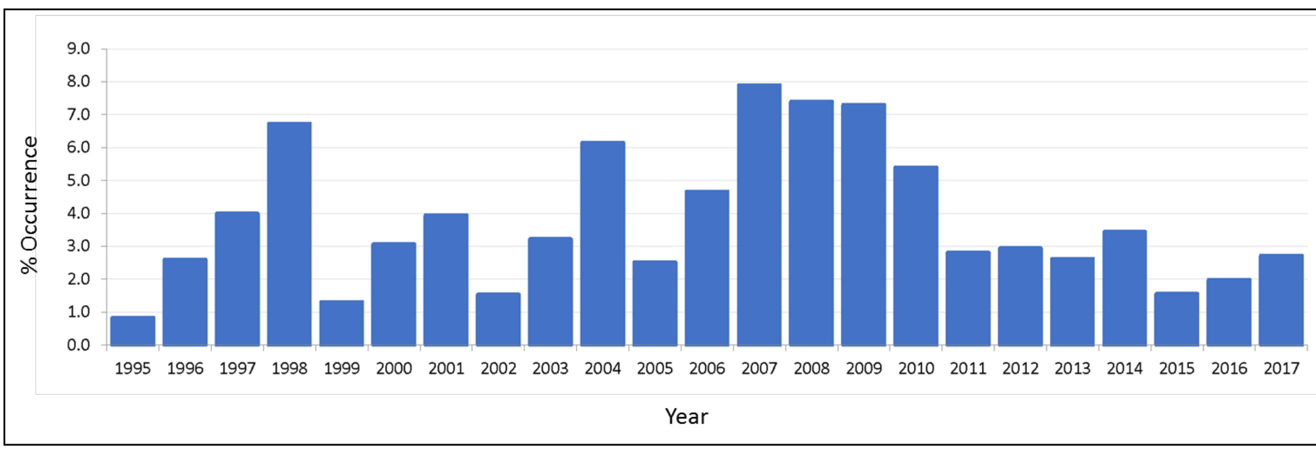

Figure 34. Percent occurrence of Johnson's seagrass in Indian River Lagoon as measured from 35 fixed-site transect surveys.

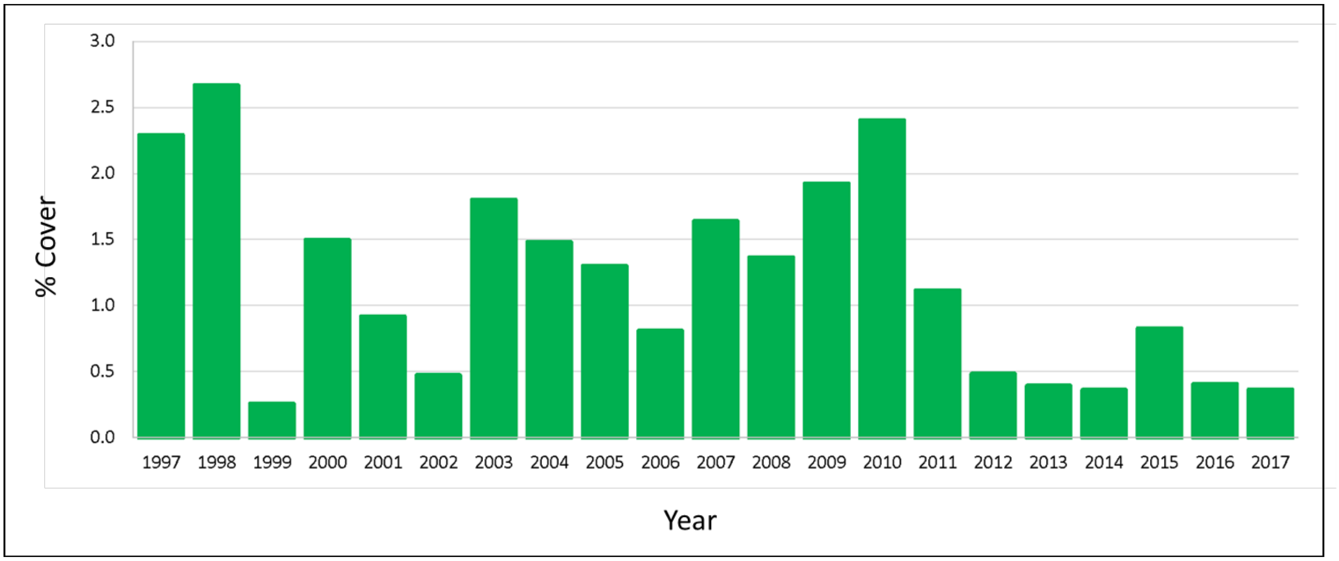

Figure 35. Percent coverage of Johnson's seagrass at 36 fixed-site transect surveys conducted each year.

In areas in which long-term poor water and sediment quality exist, Johnson's seagrass appears to occur in relatively higher abundance, perhaps due to the inability of the larger species to thrive. Like in the southern range, Johnson's seagrass appears to be out-competed in seagrass habitats where environmental conditions permit the larger seagrass species to thrive (Kenworthy 1997b; Virnstein et al. 1997). When the larger, canopy-forming species are absent, Johnson's seagrass can grow throughout the full seagrass depth range of the Indian River Lagoon (NMFS 2007a; Virnstein and Hall 2009).

4.1.7.4 Threats

A wide range of activities, many funded authorized or carried out by federal agencies, have and will continue to affect Johnson's seagrass. The Recovery Team determined that the most significant threat to the species is the destruction, modification, or curtailment of its habitat through water management practices and stochastic environmental events, especially as it relates to salinity and algal blooms. Some studies (e.g., (Dawes et al. 1989)) have shown that Johnson's seagrass has a wide tolerance for salinity; however, experiments have shown reduced photosynthesis and increased mortality at low salinities (<10 psu [practical salinity units]) (Torquemada et al. 2005). Longer duration mesocosm experiments have resulted in 100% mortality of Johnson's seagrass after 10 days at salinities <10 psu (Kahn and Durako 2008). Given that it is not uncommon for salinities to decline below 15 to 20 psu in its range (Steward et al. 2006) and that a number of natural and human-related factors can affect salinity throughout its range, the Recovery Team identified reduced salinity as a potential significant threat to the species because the potential for long-term mortality over a large scale could counteract the life history strategy the species uses to persist in the face of numerous, ongoing, environmental impacts.

Stochastic events such as hurricanes, prolonged rainfall events, and algal blooms are an ongoing threat to seagrasses, including Johnson's seagrass. Through these events, water quality is impacted in terms of salinity, nutrients, and/or transparency. Hurricanes have the potential to not only affect water quality (i.e., salinity, clarity) but can also cause seagrass burial or sediment erosion during passage. Hurricane Mathew passed along the east coast of Florida in late 2016 and is at least partly responsible for the declines in southern Indian River Lagoon seagrass coverage measured in 2017 (Kahn 2019).

High nutrient concentrations can lead to both micro- and macro- algae blooms under certain environmental conditions. Algal blooms are a source of ecological disturbance that alter water quality, bottom-dwelling species composition, and patterns of primary productivity through hypoxia (low DO) and shading (reduced sunlight penetration into the water column). The frequency and duration of bloom events dictate the impact on seagrasses. Persistent algal blooms throughout the range of Johnson's seagrass have resulted in seagrass declines over the past decade (Kahn 2019; Lirman et al. 2016; Morris et al. 2018; Orlando et al. 2016).

In previous reviews, including the critical habitat listing rule and the 2002 Recovery Plan, several additional factors were considered threats: (1) dredging and filling, (2) construction and shading from in-and over-water structures, (3) propeller scarring and anchor mooring, (4) trampling, (5) storms, and (6) siltation. In reviewing all information available since the original listing, the Recovery Team conducted assessments of each of these factors and has been unable to confirm that any of these pose a significant threat to the persistence and recovery of the species. A brief discussion of these factors follows.

The dredging of bottom sediments to maintain, or in some cases create, inlets, canals, and navigation channels may affect seagrasses by direct removal, light limitation due to turbidity, and burial from sedimentation. The disturbance of sediments can also destabilize the benthic community and resuspend nutrients, which could result in over-enrichment and/or reduce DO levels. Altering benthic topography or burying the plants may remove them from the photic zone and the altered shape and depth of the bottom within the dredged footprint may prevent future growth.

The construction of docks, marinas, and bridges can impact Johnson's seagrass through direct removal of the plants but also indirectly through habitat effects (e.g., shading and increased turbidity). Similar to dredging, installation of piles for docks or bridges can result in increased turbidity that can negatively impact water transparency over short durations. Installed piles also replace the stable, unconsolidated bottom sediments essential for the species preventing future growth in the footprint of the piles. Completed structures can have long-term effects on the availability of habitat for the species in the surrounding area because of the shade they produce. While shading does not affect water transparency directly, it does affect the amount and/or duration of sunlight that can reach the bottom. The threat posed by dock, marina, and bridge construction is especially apparent in coastal areas where Johnson's seagrass is found.

Propeller scarring and improper anchoring are known to adversely affect seagrasses (Kenworthy et al. 2002; Sargent et al. 1995). These activities can severely disrupt the benthic habitat by uprooting plants, severing rhizomes, destabilizing sediments, and significantly reducing the viability of the seagrass community. Indirect effects associated with motor vessels include turbidity from operating in shallow water, dock construction and maintenance, marina expansion, and inlet maintenance dredging. These activities and impacts are likely to increase with predicted increases in boating activity in Florida (NMFS 2007a). There are a number of local, state, and federal statutes to protect seagrasses from damage due to vessel impacts, and a number of conservation measures, including the designation of vessel control zones, signage, mooring fields, and public awareness campaigns, are directed at minimizing vessel damage to seagrasses. Despite these efforts, vessel damage can have significant local and small-scale (1 m^2 to 100 m^2) impacts on seagrasses (Kirsch et al. 2005), but there is no direct evidence that these small-scale local effects are so widespread that they are a threat to the persistence and recovery of Johnson's seagrass.

Trampling of seagrass beds, a secondary effect of recreational boating, also disturbs seagrass habitat but is a lesser concern. Trampling damages seagrasses by pushing leaves into the sediment and crushing or breaking the leaves and rhizomes. Since the designation of critical habitat, however, there have been no documented observations or reports of damage by trampling, and if there were, effects would be small-scale and local. Therefore, the Recovery Team determined that trampling does not constitute a significant threat to the survival or recovery of Johnson's seagrass.

Large-scale weather events such as tropical storms and hurricanes, while often generating runoff conditions that decrease water quality, also produce conditions (wind setup and abrupt water elevation changes) that can increase flushing rates. The effects of storms can be complex. There are several specifically documented storm effects on seagrasses: (1) scouring and erosion of sediments; (2) erosion of seeds and plants by waves, currents, and surge; (3) burial by shifting sand; (4) turbidity; and (5) discharge of freshwater, including inorganic and organic constituents in the effluents (Steward et al. 2006). Storm effects may be chronic, e.g., due to seasonal weather cycles, or acute, such as the effects of strong thunderstorms or tropical cyclones. Studies have demonstrated that healthy, intact seagrass meadows are generally resistant to physical degradation from severe storms, whereas damaged seagrass beds may not be as resilient (Fonseca et al. 2000; Whitfield et al. 2002). One post-hurricane study of Johnson's seagrass in Indian River Lagoon indicates that, while the species may temporarily decline after a storm event, it can return quickly under the right conditions (Virnstein and Morris 2007a). Furthermore, despite evidence of longer-term reductions in salinity, increased water turbidity, and increased water color associated with higher than average precipitation in the following spring, there was no evidence of long-term chronic impacts to seagrasses and no direct evidence of damage to Johnson's seagrass as a result of the hurricane that could be considered a threat to the survival of the species (Steward et al. 2006).

Silt derived from adjacent land and shoreline erosion, river and canal discharges, inlets, and internally re-suspended materials can lead to the accumulation of material on plant leaves causing light deprivation. Deposition of silt can also lead to the burial of plants, accumulation of organic matter, and the formation of anoxic sediments. Johnson's seagrass grows in a wide range of environments, including those that are exposed to siltation from all the potential sources. Documentation of the direct effects of siltation on seagrasses is generally unavailable, though it has been experimentally simulated in laboratory environments (W.J. Kenworthy, NOAA Center for Coastal Fisheries and Habitat Research, NOS, Beaufort, North Carolina, unpublished data). These experiments indicated that siltation resulting in complete burial for 12 days may cause mortality of Johnson's seagrass. In general, the effects of siltation are localized and not widespread and are not likely to threaten the survival of the species.

In addition to the 6 factors discussed above, we also consider the effects of altered water quality on Johnson's seagrass. Availability of light is one of the most significant environmental factors affecting the survival, growth, and distribution of seagrasses (Abal et al. 1994; Bulthuis 1983; Dennison 1987; Kenworthy and Fonseca 1996). Water quality and the penetration of light are affected by turbidity (suspended solids), color, nutrients, and chlorophyll, and are major factors controlling the distribution and abundance of sea grasses (Dennison 1987; Kenworthy and Fonseca 1996; Kenworthy and Haunert (editors) 1991). Increases in color and turbidity values throughout the range of Johnson's seagrass are generally caused by high flows of freshwater discharged from water management canals, which can also reduce salinity. Wastewater and storm water discharges, land runoff, and subterranean sources are also causes of increased turbidity. Degradation of water quality due to increased land use and poor water management practices continues to threaten the welfare of seagrass communities. Declines in water quality are likely to worsen, unless water management and land use practices can curb or eliminate freshwater discharges and minimize inputs of sediments and nutrients. A nutrient-rich environment caused by inorganic and organic nitrogen and phosphorous loading via urban and agricultural runoff stimulates increased algal growth that may smother or shade Johnson's seagrass and diminish the oxygen content of the water. Low oxygen conditions have a demonstrated negative impact on seagrasses and associated communities.

4.1.7.5 Climate Change Effects on Seagrasses

Here, we consider the possible effects of climate change (i.e., rising temperatures and sea levels) on seagrasses in general and on Johnson's seagrass in particular. Earth's climate is projected to warm between 2° and 4°C by 2100, and similar projections have been made for marine systems (Sheppard and Rioja-Nieto 2005). At the margins of temperate and tropical bio-regions and within tidally-restricted areas where seagrasses are growing at their physiological limits, increased temperatures may result in losses of seagrasses and/or shifts in species composition (Short et al. 2007). The response of seagrasses to increased water temperatures will depend on the thermal tolerance of the different species and their optimum temperature for photosynthesis, respiration, and growth (Short and Neckles 1999). With future climate change and potentially warmer temperatures, there may be a 1-5 m rise in the seawater levels by 2100 when taking into account the thermal expansion of ocean water and melting of ocean glaciers. Rising sea levels may adversely impact seagrass communities due to increases in water depth above present meadows, thus reducing available light. Climate change may also reduce light by shifting weather patterns to cause increased cloudiness. Changing currents may cause erosion, increased turbidity and seawater intrusions higher up on land or into estuaries and rivers, which could increase landward seagrass colonization (Short and Neckles 1999). A landward migration of seagrasses with rising sea levels is a potential benefit, so long as suitable substrate is available for colonization.

It is uncertain how Johnson's seagrass will adapt to rising sea levels and temperatures. Much depends on how much and how quickly temperatures increase. For example, Johnson's seagrass that grows intertidally (e.g., in some parts of the Lake Worth Lagoon) may be affected by a slight change in temperature (since it may already be surviving under less than optimal conditions).

However, this may be ameliorated with rising sea levels, assuming Johnson's seagrass would migrate landward with rising sea levels and assuming that suitable substrate would be available for a landward migration. However, rising sea levels could also adversely impact seagrass communities due to increases in water depths above existing meadows reducing available light. Reduction in light availability may benefit some seagrass species (e.g., *Halophila* species) that require less light compared to the larger, canopy-forming species; therefore, much depends on the thermal tolerance of the different seagrass species and their optimum temperature for photosynthesis, respiration, and growth (Short and Neckles 1999). While sea level has changed many times during the evolutionary history of Johnson's seagrass, it is uncertain how this species will fare when considering the combined effects of rising temperatures and sea levels in conjunction with other stressors such as reduced salinity from freshwater runoff. It has been shown that evolutionary change in a species can occur within a few generations(Rice and Emery 2003), thus making it possible for seagrasses to cope if the changes occur at a rate slow enough to allow for adaptation.

5 ENVIRONMENTAL BASELINE

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and the ecosystem, within the action area, without the consequences caused by the proposed action. By regulation, the environmental baseline for a Biological Opinion refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to the listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR 402.02). The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline.

Focusing on the impacts of the activities in the action area specifically allows us to assess the prior experience and state (or condition) of the endangered and threatened individuals that occur in an action area and that will be exposed to effects from the actions under consultation. This is important because, in some phenotypic states or life history stages, listed individuals will commonly exhibit, or be more susceptible to, adverse responses to stressors than they would be in other states, stages, or areas within their distributions. The same is true for localized populations of endangered and threatened species; the consequences of changes in the fitness or performance of individuals on a population's status depends on the prior state of the population. However, due to the breadth of the action area for this consultation, much of the information provided in Section 4 of this Opinion (*Status of the Species*) of this Opinion already describes past and ongoing impacts to the species within the action area. Therefore, this section will discuss factors affecting the species within the action area that were not considered (or not described in sufficient depth) elsewhere in this Opinion.

Since coral species and Johnson's seagrass differ from other ESA-listed species in that they are sessile (as opposed to mobile), when necessary, we will discuss baseline considerations unique to corals and Johnson's seagrass, in a separate sections below.

5.1 Status of Species in the Action Area

As stated in Section 2.8 of this Opinion (Action Area), the proposed action occurs in the Atlantic Ocean from North Carolina to the southern tip of the Florida Keys and the U.S. Caribbean.

Sea Turtles
Sea turtles (green [NA and SA DPS]), Kemp's ridley, hawksbill, leatherback, and loggerhead (NWA DPS) occurring in the action area are all highly migratory. Given the large size of the action area, the status of the 5 species (or DPS where applicable) of sea turtles in the action area, as well as the threats to these species, are best reflected in their range-wide statuses and supported by the species accounts in Section 4.1.1 of this Opinion (Status of Species).

Atlantic and Shortnose Sturgeon
Atlantic sturgeon (including all 5 DPSs: New York Bight, Chesapeake Bay, Carolina, and SA DPSs, and the Gulf of Maine DPS) and shortnose sturgeon may be found in the action area from North Carolina to Northern Florida. Given the large size of the action area, the status of sturgeon in the action area, as well as the threats to these DPSs, is supported by the species accounts in the Status of the Species sections of this Opinion in Section 4.1.2 for Atlantic sturgeon and their DPSs and in Section 4.1.3 for shortnose sturgeon.

Giant Manta Ray
Giant manta ray may be found throughout the action area throughout the year. Given the large size of the action area, the status of the species in the action area, as well as the threats to these species, are best reflected in their range-wide statuses and supported by the species accounts in Section 4.1.4 of this Opinion (Status of Species).

Smalltooth Sawfish
Smalltooth sawfish may be found in Florida within the action area throughout the year. However, juvenile sawfish are closely associated with mangrove shorelines and are only likely to be found in the action area in limited numbers in the Florida Keys. The status of smalltooth sawfish in the action area, as well as the threats to this species, is supported by the species account in Section 4.1.5 of this Opinion (Status of the Species).

ESA-listed corals
ESA-listed corals within the action area are found in South Florida and the U.S. Caribbean. The status of these species in the action area, as well as the threats to this species, is supported by the species account in Section 4.1.6 of this Opinion (Status of the Species).

Johnson's Seagrass
The entire range of Johnson's seagrass occurs within the action area. Therefore, the status of the species in the action area, as well as the threats to these species, are best reflected in their range-

wide statuses and supported by the species accounts in Section 4.1.7 of this Opinion(Status of Species).

5.2 Climate Change

The *2014 Assessment Synthesis Report* from the Working Groups on the Intergovernmental Panel on Climate Change concluded climate change is unequivocal (Intergovernmental Panel on Climate Change 2014). The report concludes oceans have warmed, with ocean warming the greatest near the surface (e.g., the upper 75 m [246.1 ft] have warmed by 0.11°C per decade over the period 1971 through 2010) (Intergovernmental Panel on Climate Change 2014). The Atlantic Ocean appears to be warming faster than all other ocean basins except perhaps the southern oceans (Cheng et al. 2017). In the western North Atlantic Ocean surface temperatures have been unusually warm in recent years (Blunden and Arndt (editors) 2016). A study by Polyakov et al. (2009), suggests that the North Atlantic Ocean overall has been experiencing a general warming trend over the last 80 years of 0.031±0.0006°C per decade in the upper 2,000 m (6,561.7 ft) of the ocean. The *Fourth National Climate Assessment* confirmed that the Atlantic and Gulf coasts in particular are facing above-average risks to ocean and coastal infrastructure (U.S. Global Change Research Program 2018). Also highlighted were rising water temperatures, ocean acidification, retreating arctic sea ice, sea level rise, high-tide flooding, coastal erosion, higher storm surge, and heavier precipitation events as key threats to the nation's oceans and coasts (U.S. Global Change Research Program 2018). Climate change is expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, tropical storms, heat waves, and droughts (Intergovernmental Panel on Climate Change 2014; U.S. Global Change Research Program 2018). Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Ocean acidity has increased by 26% since the beginning of the industrial era (Intergovernmental Panel on Climate Change 2014) and this rise has been linked to climate change.

Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure in the action area (Evans and Bjørge 2013; Intergovernmental Panel on Climate Change 2014; Kintisch 2006; Learmonth et al. 2006; MacLeod et al. 2005; McMahon and Hays 2006; Robinson et al. 2005). Marine species' ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). For example, increased temperatures may result in shifts in seagrass species composition (Short et al. 2007), which would affect the seagrasses themselves as well as the marine species that rely on seagrasses for forage and refuge. Though predicting the precise consequences of climate change on marine species is difficult (Simmonds and Isaac 2007), recent research has indicated a range of consequences already occurring (U.S. Global Change Research Program 2018).

Other examples include the McMahon and Hays (2006) study that found increased ocean temperatures are expanding the distribution of leatherback turtles into more northern latitudes in the Atlantic Ocean. On the opposite end of the spectrum, sessile species (corals and seagrasses, for example) are unable to expand their ranges or leave certain areas to find more suitable habitat, making it more difficult for these species to adapt to warming temperatures.

5.3 Stochastic Events

Stochastic events (e.g., hurricanes, winter storms, storm surge/inundation, earthquakes, coral bleaching events/die-offs, harmful algal blooms/red tide, etc.) may be infrequent individually, but cumulatively are common throughout the action area. These events are, by definition, unpredictable, thus their effect on the survival and recovery of listed species and designated critical habitat is also unpredictable. They have the potential to impede the survival and recovery of species directly (i.e., animals die as a result of these events) or indirectly (i.e., habitat – especially critical habitat – is damaged as a result of these events).

Corals and seagrasses can experience direct or indirect impacts due to stochastic events to which mobile species are not as susceptible. Direct impacts could include displacement, uprooting, bleaching, and desiccation, and indirect impacts could include sedimentation, limited access to sunlight, and introduction of new species expanding their ranges to the area. Monitoring data from the U.S. Virgin Islands Territorial Coral Reef Monitoring Program indicate that a 2005 coral bleaching event in the Caribbean caused the largest documented loss of coral in the U.S. Virgin Islands since coral monitoring data have been available, with a decline of at least 50% of coral cover in waters less than 25 m deep (Smith et al. 2011). The same bleaching event caused bleaching of 20% of elkhorn and 75% of staghorn corals across 12 monitored locations in Puerto Rico, and almost 100% of staghorn colonies suffered partial to complete mortality near Culebra Island (Garcia-Sais et al. 2008). Hurricanes and large coastal storms could also significantly harm staghorn coral. Due to its branching morphology, it is especially susceptible to breakage from extreme wave action and storm surges. Hurricanes are also sometimes beneficial – if they do not result in heavy storm surge – during years with high sea surface temperatures, as they lower the temperatures, thereby providing fast relief to corals during periods of high thermal stress (Heron et al. 2008). However, major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs. Hurricanes can also cause flooding that an affect the water quality in sturgeon spawning rivers, as was documented during Hurricane Florence discussed in Section 4.1.2 of this Opinion. According to the NOAA Historical Hurricane Tracks website, from 1859 to 2013, approximately, 29 hurricanes or tropical storms impacted the area within 20 nautical miles of Fort Lauderdale, Florida.

Prolonged cold water temperatures can cause “cold stunning” and mortalities of sea turtles and fish. In January 2010, an unusually long bout of cold weather in Florida led to a statewide sea turtle cold-stunning event.[60] The 2010 cold snap in Florida also caused mortalities of smalltooth sawfish (Poulakis et al. 2011b).

In 2017, Hurricanes Irma and Maria caused significant damage to waters around Puerto Rico and the U.S. Virgin Islands, and parts of the state of Florida. The damage included:

- uprooted mangrove habitat used by smalltooth sawfish and Nassau grouper;
- displaced coral colonies (affecting ESA-listed coral species and serving as habitat for Nassau grouper and sea turtles);

[60] http://myfwc.com/research/wildlife/sea-turtles/mortality/about/

- reduced salinity in coastal habitats, reduced oxygen in freshwater streams, and increased turbidity in both coastal and freshwater habitats used by Atlantic and shortnose sturgeon;
- increased sedimentation, affecting many different habitats utilized by ESA-listed species in the region,

Stochastic events will likely continue to be common occurrences throughout the action area.

5.4 Vessel Strike

Vessels have the potential to affect sea turtles, sturgeon, giant manta ray, and marine mammals through strikes, sound, and disturbance associated with their physical presence and can affect listed corals and Johnson's seagrass through propeller scarring, propeller wash, and accidental groundings, as discussed in Section 3 of this Opinion. Vessel strikes are considered a serious and widespread threat to ESA-listed species. This threat is increasing as commercial shipping lanes cross important habitats (Swingle et al. 1993; Wiley et al. 1995). As vessels become faster and more widespread, an increase in vessel interactions with ESA-listed species is to be expected.

5.5 Fisheries

Fisheries constitute an important and widespread use of the ocean resources throughout the action area. Fisheries can adversely affect ESA-listed species and critical habitat. Direct effects of fisheries interactions on ESA-listed species include entanglement, entrapment, and ingestion of gear, which can lead to fitness consequences or mortality as a result of injury or drowning. Indirect effects include reduced prey availability, including overfishing of targeted species, and destruction of habitat. Use of mobile fishing gear, such as bottom trawls, disturbs the seafloor and reduces structural complexity. Indirect impacts of trawls include increased turbidity, alteration of surface sediment, removal of prey (leading to declines in predator abundance), removal of predators, ghost fishing (i.e., lost fishing gear continuing to ensnare fish and other marine animals), and generation of marine debris. Lost gill nets, purse seines, and long-lines may foul and disrupt bottom habitats and injure coral species, and have the potential to entangle or be ingested by marine mammals, sea turtles, and large fish species. Fisheries can also have a profound influence on fish populations. In a study of retrospective data, Jackson et al. (2001) concluded that ecological extinction caused by overfishing precedes all other pervasive human disturbance of coastal ecosystems, including pollution and anthropogenic climatic change.

5.5.1 Fisheries Interactions

Within the action area, both recreational and commercial fisheries occur in state/territorial and federal waters. Globally, 6.4 million tons of fishing gear is lost in the oceans every year (Wilcox et al. 2015). Lost traps and pots ("ghost gear") often wind up settling in and around coral reefs and habitat suitable for coral recruitment (hardbottom substrate) and is becoming better-documented as a source of human-caused mortality in sea turtles (Barnette 2017). Materials entangled tightly around a body part may cut into tissues, enable infection, and severely compromise an individual's health (Derraik 2002). Entanglements also make animals more vulnerable to additional threats (e.g., predation and vessel strikes) by restricting agility and swimming speed. The majority of ESA-species that die from entanglement in fishing gear likely sink at sea rather than strand ashore, making it difficult to accurately determine the extent of such mortalities. Reductions in fish populations, whether natural or human-caused from activities such as fishing, may affect the survival and recovery of ESA-listed species that feed on these fish including smalltooth sawfish. A reduction in herbivorous fish species that control algal growth on reefs can also adversely affect ESA-listed corals, as discussed in Section 4.1.6.1 of this Opinion.

Fishery interaction remains a major factor in sea turtle recovery and, frequently, the lack thereof. Wallace et al. (2010) estimated that worldwide, 447,000 sea turtles are killed each year from bycatch in commercial fisheries. NMFS (2002a) estimated that 62,000 loggerhead turtles have been killed as a result of incidental capture and drowning in shrimp trawl gear. Although sea turtle excluder devices and other bycatch reduction devices have significantly reduced the level of bycatch to sea turtles and other marine species in U.S. waters, mortality still occurs. Smalltooth sawfish, giant manta rays, and sturgeon are also caught as bycatch in fisheries.

In addition to commercial bycatch, recreational hook-and-line interactions also occur. Data compiled by the Florida Sea Turtle Stranding and Salvage Network shows that along the coast of the Southeast U.S., from 2007-2015 approximately 300 sea turtles were reported as having recreational hook-and-line interactions (Florida Sea Turtle Stranding and Salvage Network). Recreational hook-and-line interactions are also known to sturgeon, smalltooth sawfish, and giant manta ray.

Fisheries in federal waters have been the subject of multiple Section 7 consultations in the action area. These fisheries include gillnet, longline, other types of hook-and-line gear, trawl gear, and pot fisheries. As described in Section 4 of this Opinion, available information suggests that mobile ESA-listed species can be captured in these gear types when the operation of the gear overlaps with the distribution of the species. For all fisheries for which there is a federal Fishery Management Plan, or for which any federal action has been taken to manage that fishery, impacts have been evaluated under Section 7. Formal Section 7 consultations have been conducted on the following fisheries, occurring at least in part within the action area, found likely to adversely affect threatened and endangered species: Atlantic swordfish/tuna/billfish, Atlantic shark fisheries, Caribbean reef fish fishery, Caribbean spiny lobster fishery, coastal migratory pelagic, dolphin/wahoo, South Atlantic snapper-grouper, Southeast shrimp trawl, and tilefish fisheries. An ITS has been issued for the take in each of these fisheries. None of the Opinions for these fisheries concluded that the fisheries at issue were likely to jeopardize ESA-listed species or adversely modify designated critical habitat. Detailed information regarding the effects of each fishery can be found in the respective Biological Opinions.

5.5.2 Aquaculture

Aquaculture has the potential to impact protected species via entanglement and/or other interaction with aquaculture gear (i.e., buoys, nets, and lines), introduction or transfer of pathogens, increased vessel traffic and noise, impacts to habitat and benthic organisms, and water quality (Clement 2013; Lloyd 2003; Price and Morris Jr. 2013; Price et al. 2017). Current data suggest that interactions and entanglements of ESA-listed species with aquaculture gear are rare (Price et al. 2017). This may be because worldwide the number and density of aquaculture farms are low, and thus there is a low probability of interactions, or because they pose little risk to ESA-listed species. Nonetheless, given that some aquaculture gear, such as that used in off-bottom aquaculture that is starting to increase in use in the southeast, is similar to gear used in commercial fisheries, aquaculture may cause impacts similar to those already mentioned in the fisheries discussion above (bycatch, fisheries interactions, etc.). There are very few reports of ESA-listed species interactions with aquaculture gear in the U.S. Atlantic Ocean, although it is not always possible to determine if the gear animals become entangled in is from aquaculture or commercial fisheries (Price et al. 2017). Informal consultations completed on aquaculture (NMFS 2015b) require vertical lines for off-bottom aquaculture be non-entangling and placement on materials for aquaculture avoid critical habitat features and important biological resources.

5.6 Pollution

Within the action area, pollution poses a threat to ESA-listed species. Pollution can come in the form of marine debris, pesticides, contaminants, and hydrocarbons.

5.6.1 Marine Debris

Marine debris is an ecological threat that is introduced into the marine environment through ocean dumping, littering, or hydrologic transport of these materials from land-based sources (Gallo et al. 2018). Even natural phenomena, such as tsunamis and continental flooding, can cause large amounts of debris to enter the ocean environment (Watters et al. 2010). Marine debris has been discovered to be accumulating in gyres throughout the oceans. Marine mammals, sea turtles, fish (including elasmobranchs) and corals often become entangled in marine debris, including fishing gear (Baird et al. 2015). Despite debris removal and outreach to heighten public awareness, marine debris in the environment has not been reduced (National Research Council 2008) and continues to accumulate in the ocean and along shorelines within the action area.

Marine debris affects marine habitats and marine life worldwide, primarily by entangling or choking individuals that encounter it (Gall and Thompson 2015). Entanglement in marine debris can lead to injury, infection, reduced mobility, increased susceptibility to predation, decreased feeding ability, fitness consequences, and morality for ESA-listed species in the action area. Entanglement can also result in drowning for air breathing marine species such as sea turtles. The ingestion of marine debris has been documented to result in blockage or obstruction of the digestive tract, mouth, and stomach lining of various species and can lead to serious internal injury or mortality (Derraik 2002). Law et al. (2010) presented a time series of plastic content at the surface of the western North Atlantic Ocean and Caribbean Sea from 1986 through 2008. More than 60% of 6,136 surface plankton net tows collected small, buoyant plastic pieces. Data on marine debris in some locations of the action area is largely lacking; therefore, it is difficult to draw conclusions as to the extent of the problem and its impacts on populations of ESA-listed species.

Ingestion of marine debris can be a serious threat to sea turtles, as discussed in Section 4.1.1.1 of this Opinion. When feeding, sea turtles (e.g., leatherback turtles) can mistake debris (e.g., tar and plastic) for natural food items, especially jellyfish, which are a primary prey. Some types of marine debris may be directly or indirectly toxic, such as oil. One study found plastic in 37% of dead leatherback turtles and determined that 9% of those deaths were a direct result of plastic ingestion (Mrosovsky et al. 2009). Plastic ingestion is very common in leatherback turtles and can block gastrointestinal tracts leading to death (Mrosovsky et al. 2009). The giant manta ray also faces threats from microplastics and marine debris and it is assumed other ESA-listed species also face similar threats. Other types of marine debris, such as discarded or derelict fishing gear and cargo nets, may entangle and drown sea turtles and entanglement may lead to death of other species like the smalltooth sawfish if the entanglement prevents them from performing essential life functions like foraging.

Plastic debris is a major concern because it degrades slowly and many plastics float. The floating debris is transported by currents throughout the oceans and has been discovered accumulating in oceanic gyres (Law et al. 2010). Additionally, plastic waste in the ocean chemically attracts hydrocarbon pollutants such as polychlorinated biphenyl and dichlorodiphenyltrichloroethane. Fish (including elasmobranchs), marine mammals, sea turtles and corals can mistakenly consume these wastes containing elevated levels of toxins instead of their prey. A recent study of corals found within the action area found that some corals actually prefer to ingest plastic over organic-free sand (Allen et al. 2017). It is expected that ESA-listed sea turtles, fish (including elasmobranchs), and corals may be exposed to marine debris over the course of the proposed action, although the risk of ingestion or entanglement and the resulting impacts cannot be accurately predicted.

5.6.2 Pesticides and Contaminants

Exposure to pollution and contaminants have the potential to cause adverse health effects in marine species. Marine ecosystems receive pollutants from a variety of local, regional, and international sources, and their levels and sources are therefore difficult to identify and monitor (Grant and Ross 2002). Marine pollutants come from multiple municipal, industrial, and household as well as from atmospheric transport (Garrett 2004; Grant and Ross 2002; Hartwell 2004; Iwata et al. 1993). Contaminants may be introduced by rivers, coastal runoff, wind, ocean dumping, dumping of raw sewage by boats and various industrial activities, including offshore oil and gas or mineral exploitation (Garrett 2004; Grant and Ross 2002; Hartwell 2004).

Numerous factors can affect concentrations of persistent pollutants in ESA-listed species, such as age, sex and birth order, diet, and habitat use (Mongillo et al. 2012). In sea turtles, a variety of heavy metals have been found in tissues in levels that increase with sea turtle size (Anan et al. 2001; Barbieri 2009; Fujihara et al. 2003; García-Fernández et al. 2009; Gardner et al. 2006; Godley et al. 1999; Saeki et al. 2000; Storelli et al. 2008). Cadmium has been found in leatherback turtles at the highest concentration compared to any other marine vertebrate (Caurant et al. 1999; Gordon et al. 1998). Newly emerged hatchlings have higher concentrations than are present when laid, suggesting that metals may be accumulated during incubation from surrounding sands (Sahoo et al. 1996).

Sea turtle tissues have been found to contain organochlorines and many other persistent organic pollutants. Polychlorinated biphenyl (a compound found in engine coolants) concentrations in sea turtles are reportedly equivalent to those in some marine mammals, with liver and adipose levels of at least one congener being exceptionally high (PCB 209: 500-530 ng/g wet weight (Davenport et al. 1990b; Oros et al. 2009). PCBs have been found in leatherback turtles at concentrations lower than expected to cause acute toxic effects, but might cause sublethal effects on hatchlings (Stewart et al. 2011).

Organochlorines have the potential to suppress the immune system of loggerhead turtles and may affect metabolic regulation (Keller et al. 2006; Keller et al. 2005; Oros et al. 2009). The contaminants can cause deficiencies in endocrine, developmental, and reproductive health (Storelli et al. 2007) and are known to depress immune function in loggerhead turtles (Keller et al. 2006). Females from sexual maturity through reproductive life should have lower levels of contaminants than males because contaminants are shared with progeny through egg formation. Exposure to sewage effluent may also result in green turtle eggs harboring antibiotic resistant strains of bacteria (Al-Bahry et al. 2009).

Water pollution may be linked to reduced health in species such as potentially causing the fibropapilloma virus that kills many turtles each year (Foley et al. 2005). In the case of fibropapilloma, it is pollution that may be either causing or making sea turtles more susceptible to disease by weakening their immune systems.

Point source discharge (e.g., industrial discharge or overflow of pollutant containment facilities during flooding) or nonpoint source discharge (e.g., agriculture and residential runoff) may also affect water quality. For example, the Cape Fear River continues to have decreased DO levels assumed to result from non-point source and point source discharges into the river. This was further exacerbated by the overflow of coal-ash containment ponds containing heavy metals including arsenic, lead, mercury, and chromium and hog farm containment ponds filled with hog urine and feces that breached during Hurricane Florence in 2018. Mass fish kills occurred in the Cape Fear River either from the large amount of fresh water in the river from the Hurricane and/or the contaminants released.

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, are known to have adverse effects on ESA-listed corals and Johnson's seagrass, as discussed in the status of the species sections in Section 4 of this Opinion. Lapointe et al. (2004) directly linked wastewater discharges in the Florida Keys with adverse effects to the nearby coral reef communities. Nutrients, contaminants, and sediment from point and non-point sources create an unfavorable environment for reproduction and growth of corals by promoting overgrowth of hard substrate by algae or the buildup of sediment layers that prohibit coral settlement.

5.6.3 Hydrocarbons

There has never been a large-scale oil spill in the action area, but numerous small-scale vessel spills likely occur. A nationwide study examining vessel oil spills from 2002 through 2006 found that over 1.8 million gallons of oil were spilled from vessels in all U.S. waters (Dalton and Jin 2010). In this study, "vessel" included numerous types of vessels, including barges, tankers, tugboats, and recreational and commercial vessels, demonstrating that the threat of an oil spill can come from a variety of boat types. Below we review the effects of oil spills on ESA-listed species more generally. Much of what is known comes from studies of large oil spills. Since no information exists on the effects of oil spills within the action area, we must use information from other oil spill events outside the action area to inform our analysis of potential effects to species and critical habitat within the action area.

Exposure to hydrocarbons released into the environment via oil spills and other discharges pose risks to marine species. For example, Oil can also be hazardous to sea turtles, with fresh oil causing significant mortality and morphological changes in hatchlings, but aged oil having no detectable effects (Fritts and McGehee 1982). For example, the *Deepwater Horizon* oil spill extensively oiled vital foraging, migratory, and breeding habitats of sea turtles throughout the northern Gulf of Mexico (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). *Sargassum* habitats, benthic foraging habitats, surface and water column waters, and sea turtle nesting were all affected by the *Deepwater Horizon* oil spill. Sea turtles may have been exposed to *Deepwater Horizon* oil in contaminated habitats, through breathing oil droplets, oil vapors, and smoke, by ingesting oil-contaminated water and prey, and through maternal transfer of oil compounds to developing embryos. Translocation of eggs from the Gulf of Mexico to the Atlantic Ocean coast of Florida resulted in the loss of sea turtle hatchlings. High numbers of small oceanic and large sea turtles are estimated to have been exposed to oil resulting from the *Deepwater Horizon* oil spill due to the duration and large footprint of the oil spill. It was estimated that as many as 7,590 large juvenile and adult sea turtles (Kemp's ridley, loggerhead, and unidentified hardshelled sea turtles) and up to 158,900 small juvenile sea turtles (hawksbill, Kemp's ridley, loggerhead, and hardshelled sea turtles not identified to species) were killed by the *Deepwater Horizon* oil spill. Small juveniles were affected in the greatest numbers and suffered a higher mortality rate than large sea turtles. Leatherback turtle foraging and migratory habitat was also affected and though impacts to leatherback turtles were unquantified, it is likely some died as a result of the *Deepwater Horizon* oil spill and spill response (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016; NMFS and USFWS 2013).

As noted above, to our knowledge the past and present impacts of oil spills on ESA-listed species within the action area are limited to those associated with small-scale vessel spills. Nevertheless, we consider the documented effects of oil spills outside the action area, such as the *Deepwater Horizon* oil spill, examples of the possible impacts that an oil spill can have on ESA-listed species such as those described for sea turtles. Non-mobile species such as ESA-listed coral and Johnson's seagrass would be expected to have even more detrimental effects if a spill would occur where they are present since they would not be able to avoid such an interaction.

5.7 Aquatic Nuisance Species

Aquatic nuisance plants and animals introduced into new habitats throughout the U.S. and other areas of the world that produce harmful impacts on aquatic ecosystems and native species (http://www.anstaskforce.gov). They are also referred to as invasive, alien, or non-indigenous species. Introduction of these species is cited as a major threat to biodiversity, second only to habitat loss (Wilcove et al. 1998). A variety of vectors are thought to have introduced non-native species including, but not limited to aquarium and pet trades, recreation, and ballast water discharges from ocean-going vessels. Common impacts of invasive species are alteration of habitat and nutrient availability, as well as altering species composition and diversity within an ecosystem (Strayer 2010). Shifts in the base of food webs, a common result of the introduction of invasive species, can fundamentally alter predator-prey dynamics up and across food chains

(Moncheva and Kamburska 2002), potentially affecting prey availability and habitat suitability for ESA-listed species.

5.8 Disease

Diseases discussed in Section 4 of this Opinion (Status of the Species) can cause a decline in species populations. For example, green sea turtles are susceptible to natural mortality from Fibropapillomatosis disease, as discussed in Section 4.1.1.2.4 of this Opinion.

As discussed in Section 4.1.6.1 of this Opinion, coral diseases are a common and significant threat affecting most or all coral species and regions. Since 2014, the Florida Reef Tract has been experiencing the most widespread and lethal coral disease outbreak in the world. While disease outbreaks are not uncommon, the current outbreak is unique due to the high number of coral species affected across a large portion of the Florida Reef Tract. Additionally, it has been documented in 9 locations throughout the Caribbean. As of right now, 23 of the 45 coral species present in Florida are thought to be affected, including the ESA-listed boulder star coral, mountainous star coral, lobed star coral, pillar coral, and rough cactus coral. A key factor is that this disease exhibits high rates of transmission and mortality once a coral is infected (full colony mortality observed within weeks to months). This disease outbreak was first reported in the fall 2014 near Key Biscayne, Miami-Dade County. By 2019 the disease had reached approximately 15 miles west of Key West in the lower Keys (Figure 36). For more information, refer to the Florida Department of Environmental Protection's coral disease website, which is available at the following link: https://floridadep.gov/rcp/coral/content/florida-reef-tract-coral-diseaseoutbreak.

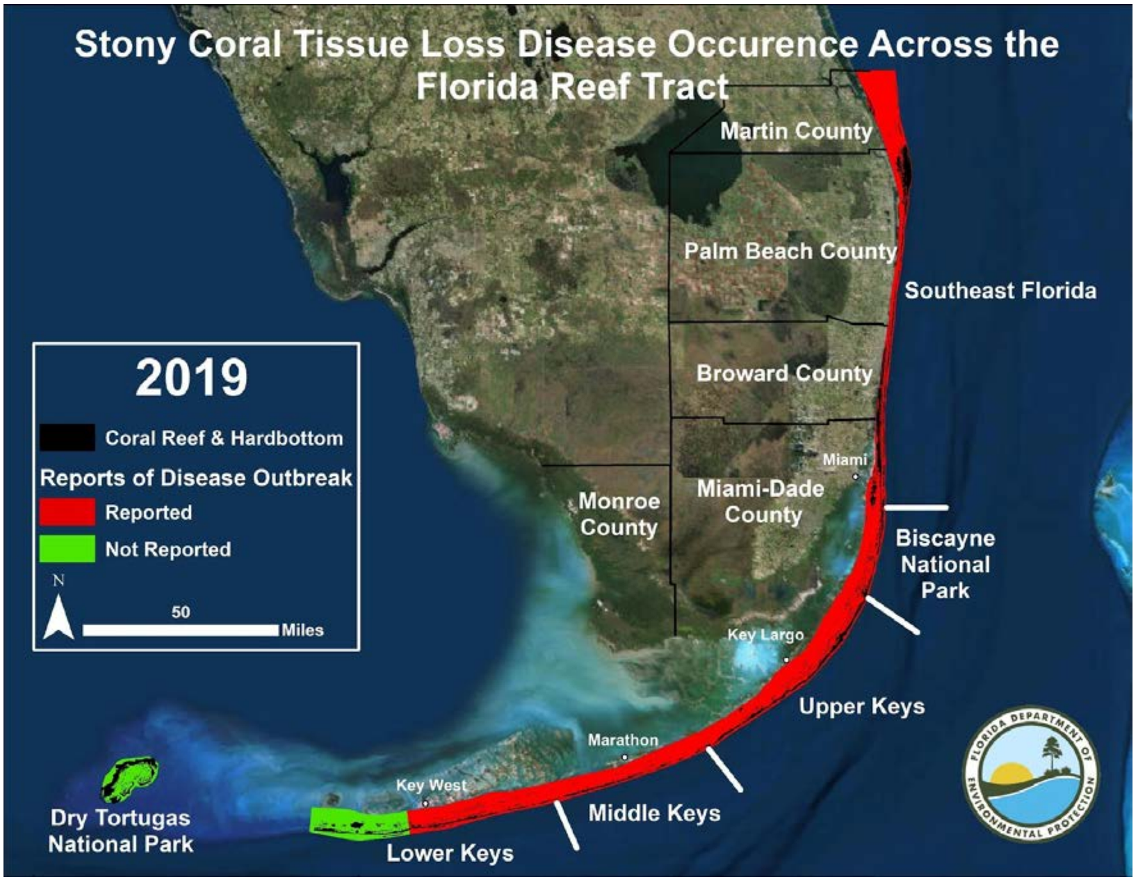

Figure 36. Extent of coral disease outbreak across the Florida reef tract.

The degradation of coral habitat may potentially lead to a decrease in the abundance of fish that clean other species such as the giant manta ray, as discussed in Section 4.1.4.4 of this Opinion, leading to an overall reduction in the number of cleaning stations available to manta rays within these habitats. Decreased access to cleaning stations may negatively affect the fitness of giant manta rays by hindering their ability to reduce parasitic loads and dead tissue, which could lead to increases in diseases and declines in reproductive fitness and survival rates.

5.9 Sound

The ESA-listed species that occur in the action area are regularly exposed to several sources of anthropogenic sounds. These include, but are not limited to, maritime activities (vessel traffic), aircraft, exploration and research (seismic surveys), and marine construction (dredging). Anthropogenic sound in the action areas may be generated by commercial and recreational vessels, sonar, aircraft, seismic surveys, in-water construction activities, wind farms, military activities, and other human activities. These activities occur to varying degrees throughout the year. The scientific community recognizes the addition of anthropogenic sound to the marine environment as a stressor that can possibly harm marine animals or significantly interfere with their normal activities (National Research Council 2005). The species considered in this Opinion may be impacted by anthropogenic sound in various ways. Once detected, some sounds may produce a behavioral response, including but not limited to, changes in habitat to avoid areas of higher sound levels, changes in diving behavior, or, for cetaceans, changes in vocalization (Marine Mammal Commission 2007).

Many researchers have described behavioral responses of marine mammals to sounds produced by boats and vessels, as well as other sound sources such as helicopters and fixed-wing aircraft, and dredging and construction (reviewed in (Gomez et al. 2016); and (Nowacek et al. 2007). Most observations have been limited to short-term behavioral responses, which included avoidance behavior and temporary cessation of feeding, resting, or social interactions; however, in terrestrial species habitat abandonment can lead to more long-term effects, which may have implications at the population level (Barber et al. 2010). Masking may also occur, in which an animal may not be able to detect, interpret, and/or respond to biologically relevant sounds. Geophysical surveys occur throughout the Atlantic Ocean, during most (if not all) times of the year. They are conducted for purposes of scientific research (e.g., to map the seafloor in a certain area, discover a shipwreck, etc.) as well as for resource exploration (e.g., oil and gas extraction). Sound is the primary method for these exploratory surveys; by choosing specific types of sound at specific frequencies and pressures, surveys can, among other things, penetrate the seafloor to look for oil or gas; map the topography of the ocean floor; identify the type of substrate on the seafloor; aid in navigation; and search for fish. While these are all types of surveys that occur throughout the action area, the ones with the potential to cause harm or injury to listed species are discussed below. Recent consultations that evaluate the effects of geophysical surveys within the action area include the use of surveying by BOEM to evaluate borrow sites in federal waters discussed in Section 2.5.4 in Appendix K under the additional consultation history information and those conducted by the U.S. Navy discussed below.

There are seismic survey activities involving towed airgun arrays that may occur within the action area. They are the primary exploration technique to locate oil and gas deposits, fault structure, and other geological hazards. These activities may produce noise that could impact ESA-listed species within the action area. These airgun arrays generate intense low-frequency sound pressure waves capable of penetrating the seafloor and are fired repetitively at intervals of 10 to 20 seconds for extended periods (National Research Council 2003). Most of the energy from the airguns is directed vertically downward, but significant sound emission also extends horizontally. In the U.S., all seismic surveys involving the use of airguns with the potential to take marine mammals are covered by incidental take authorizations under the MMPA. Those with the potential to take any other ESA-listed species are covered by ITS, which are authorized at the conclusion of formal ESA section 7 consultation. BOEM authorizes oil and gas activities in domestic waters, while the National Science Foundation and U.S. Geological Survey also funds and/or conducts these activities in domestic and foreign waters, and in doing so, consults with NMFS to ensure their actions do not jeopardize the continued existence of ESA-listed species or adversely modify or destroy designated critical habitat.

High-resolution geophysical surveys such as those considered in this Opinion (Section 3.1.8.3 of this Opinion), do not use airguns; rather, these surveys use equipment that can be towed at the surface and shallow subsurface, as well as at the seafloor. These survey methods provide more details and cover less area than airgun survey and occur regularly throughout the Atlantic, including in the action area, for scientific research (e.g., research conducted by U.S. Geological Survey) and exploration, as well as to locate specific sand borrow and mineral disposal sites (i.e., areas used by USACE and BOEM). Most geophysical survey equipment (other than airguns) produce sound different from, and at higher frequencies than, airguns, generally outside of the hearing range of the ESA-listed species in the action area and are therefore analyzed differently than airguns for potential effects to listed species.

5.10 Marine Construction

Marine construction that can occur in the action area includes drilling, dredging, mining, pile driving, cable laying, and explosions. These activities are known to cause behavioral disturbance and physical damage (National Research Council 2003). Marine construction in the action area that has and may continue to occur also includes creating artificial reefs, dock and seawall installation, and other routine coastal development.

5.10.1 Hopper Dredging, Sand Mining, and Beach Nourishment

The construction and maintenance of Federal navigation channels, sand mining ("borrow") activities, beach nourishment, and shoreline restoration/stabilization projects have been identified as sources of incidental take for sea turtles, sturgeon, coral, and Johnson's seagrass, as described in Section 3 and Section 6 of this Opinion. Dredging, mining, and beach nourishment equipment can contribute significant obstructions and artificial lighting nearshore that can disorient sea turtle hatchling leaving nesting beaches. Dredging and placement of dredged material can also generate turbidity that can result in sedimentation and burial of ESA-listed corals and Johnson's seagrass. The navigation channels, secondary channels, ports, and other areas that may be maintenance dredged under this Opinion, have been constructed pursuant to Federal authorization or permitting. The initial construction of these projects may have altered the hydrology of these areas, such as major dredging projects in sturgeon rivers. Similarly, previously authorized or permitted sand mining and placement sites may have altered habitats available for ESA-listed species. Federally permitted or federally authorized dredging, sand mining, and nourishment within the action area have been subject to subject to Section 7 consultation. These consultations have included relocation trawling activities, which have been successful at temporarily displacing sea turtles, sturgeon, giant manta ray, and smalltooth sawfish from channels and nearshore mining areas during periods when hopper dredging was imminent or ongoing.

While interactions with ESA-listed species have been reported in the past during dredging and material placement projects, and are expected to continue in the future, previous Section 7 consultation on these activities have concluded that none of these activities were likely to jeopardize the continued existence of any ESA-listed species. Examples of these types of consultations include the expansions of the Port of Miami (NMFS 2011a) and Savannah Harbor Expansion Project (NMFS 2010c). The Port of Miami project required relocation of all 31 known Acropora cervicornis colonies out of the project area to nearby suitable reef sites as a RPM. In Savannah Harbor Expansion Project, relocation trawling was used to temporarily displace sea turtles and sturgeon from channels while hopper dredging was imminent or ongoing.

5.11 Military Activities

The U.S. Navy conducts training, testing, and other military readiness activities on range complexes throughout coastal and offshore areas in the United States and on the high seas. The U.S. Navy's activities are conducted off the coast of the Atlantic Ocean and elsewhere throughout the world. The U.S. Navy's Atlantic Fleet Training and Testing range complex overlaps with the action area for the U.S. Geological Survey's seismic survey. During training, existing and established weapon systems and tactics are used in realistic situations to simulate and prepare for combat. Activities include: routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. The U.S. Navy performs testing activities to ensure that its military forces have the latest technologies and techniques available to them. The majority of the training and testing activities the U.S. Navy conducts in the action area are similar, if not identical to activities that have been occurring in the same locations for decades.

The U.S. Navy's activities produce sound and visual disturbance to marine mammals, sea turtles, and fish throughout the action area (NMFS 2015a; NMFS 2017c). Anticipated impacts from harassment due to the U.S. Navy's activities include changes from foraging, resting, milling, and other behavioral states that require low energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures. Based on the currently available scientific information, behavioral responses that result from stressors associated with these training and testing activities are expected to be temporary and will not affect the reproduction, survival, or recovery of these species. The U.S. Navy's activities constitute a federal action, and take of ESA-listed species considered for these activities have previously undergone separate ESA Section 7 consultation. Through these consultations with NMFS, the U.S. Navy has implemented monitoring and conservation measures to reduce the potential effects of underwater sound from activities on ESA-listed resources in the Atlantic Ocean. Conservation measures include employing visual observers and implementing mitigation zones during activities using active sonar and explosives.

In addition to these testing and training activities, the U.S. Navy operates Surveillance Towed Array Sensor System Low Frequency Active sonar within the action area, which utilizes low frequency sounds to detect and monitor submarines. The Surveillance Towed Array Sensor System Low Frequency Active sonar has a coherent low-frequency signal with a duty cycle of less than 20%, operating for a maximum of only 255 hours per year for each system (or 432 hours per year in the past) or a total of 10.6 days per year. This compares to an approximate 21.9 million days per year for the world's shipping industry. Thus, Surveillance Towed Array Sensor System Low Frequency Active sonar transmissions will make up a very small part of the human-caused sound pollution in the ocean.

Prior to 2017, the U.S. Navy used Surveillance Towed Array Sensor System Low Frequency Active sonar in the western and central North Pacific Ocean. However, in 2017 the U.S. Navy requested programmatic Section 7 consultation for the operation of Surveillance Towed Array Sensor System Low Frequency Active sonar from August 2017 through 2022 in the non-polar region of the world's oceans (including within the action area). The consultation was concluded in August 2017 (NMFS 2017d) and considered the U.S. Navy's Surveillance Towed Array Sensor System Low Frequency Active sonar program and operations.

5.12 Scientific Research Activities

Regulations for Section 10(a)(1)(A) of the ESA allow issuance of permits authorizing take of certain ESA-listed species for the purposes of scientific research. Prior to the issuance of such a permit, the proposal must be reviewed for compliance with Section 7 of the ESA. Scientific research permits issued by NMFS currently authorize studies of ESA-listed species in the Atlantic Ocean, some of which may occur within the action area for the proposed action. ESA-listed species have been the subject of field studies for decades. The primary objective of most of these field studies has generally been monitoring populations or gathering data for behavioral and ecological studies. Over time, NMFS has issued dozens of permits on an annual basis for various forms of "take" of listed species in the action area from a variety of research activities.

Authorized research on sea turtles includes close approach, capture, handling and restraint, tagging, blood and tissue collection, lavage, ultrasound, imaging, antibiotic (tetracycline) injections, captive experiments, laparoscopy, and mortality. Most research activities involve authorized sublethal "takes," with some resulting mortality.

There have been numerous research permits issued under the provisions of the ESA authorizing scientific research on listed species all over the world, including for research in the action area. The consultations which took place on the issuance of these ESA scientific research permits each found that the authorized research activities will not result in jeopardy to the species or adverse modification of designated critical habitat.

Additional "take" is likely to be authorized in the future as additional permits are issued. It is noteworthy that monitoring and reporting indicate that the actual number of "takes" rarely approach the number authorized. However, our analysis assumes that these "takes" will occur since they have been authorized. It is also noteworthy that these "takes" are distributed across the Atlantic Ocean. Although ESA-listed species are generally wide-ranging, we do not expect many of the authorized "takes" to involve individuals that will also be "taken" under the proposed activities.

The National Ocean Service's Office of National Marine Sanctuaries, National Park Service, USFWS, Florida Fish and Wildlife Conservation Commission, Puerto Rico Department of Natural and Environmental Resources and the U.S. Virgin Islands Department of Planning and Natural Resources all issue permits for the collection of corals and other marine species for scientific and educational purposes. Through the ESA Section 4(d) regulations promulgated by NMFS to conserve elkhorn and staghorn corals, NMFS recognized that the permit processes for these agencies are consistent with ESA Section 10 permit requirements, and an additional permit from NMFS is not required for scientific research and enhancement activities involving elkhorn and staghorn corals. Take is not prohibited for the other 5 species of ESA-listed corals and giant manta ray, which are listed as threatened, as NMFS has not promulgated a rule under ESA Section 4(d).

At present there are 27 active Section 10(a)(1)(A) permits in the action area of this Opinion. Nineteen of these are focused on sea turtles, 5 on sturgeon, 3 on smalltooth sawfish, 1 on North Atlantic right whale, and 2 multi-species looking at both sea turtles and sturgeon. Authorizations and Permits for Protected Species, https://apps.nmfs.noaa.gov/search/search.cfm, searched 10/21/2019). Each of these permits issued received an ESA Section 7 consultation.

5.13 Conservation and Recovery Actions Shaping the Environmental Baseline

5.13.1 Sea turtles

NMFS has implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for the Atlantic highly migratory species and South Atlantic snapper-grouper fisheries, TED requirements for the Southeast shrimp trawl and North Carolina flynet fisheries, mesh size restrictions in the North Carolina gillnet fishery, and area closures in the North Carolina gillnet fishery. In addition to regulations, outreach programs have been established and data on sea turtle interactions with recreational fisheries has been collected through the Marine Recreational Fishery Statistical Survey (MRFSS)/Marine Recreational Information Program.

Sea Turtle Handling and Resuscitation Techniques
NMFS published a Final Rule (66 FR 67495, Publication Date December 31, 2001) detailing handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities. Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the Final Rule. These measures help to prevent mortality of hardshell turtles caught in fishing or scientific research gear.

Outreach and Education, Sea Turtle Rescue and Rehabilitation
There is an extensive network of Sea Turtle Stranding and Salvage Network participants along the Atlantic coast who not only collect data on dead sea turtles, but also rescue and rehabilitate any live stranded sea turtles.

A Final Rule (70 FR 42508, Publication Date July 25, 2005), allows any agent or employee of NMFS, the USFWS, the USCG, or any other federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. NMFS already affords the same protection to sea turtles listed as threatened under the ESA [50 CFR 223.206(b)].

5.13.2 Sturgeon

In 1998, the Atlantic States Marine Fisheries Commission instituted a coast-wide moratorium on the harvest of Atlantic sturgeon, which is to remain in effect until there are at least 20 protected age classes in each spawning stock (anticipated to take up to 40 or more years). NMFS followed the Atlantic States Marine Fisheries Commission moratorium with a similar moratorium on the harvest of Atlantic sturgeon in federal waters. Amendment 1 to Atlantic States Marine Fisheries Commission's Atlantic Sturgeon Fishery Management Plans also includes measures for preservation of existing habitat, habitat restoration and improvement, monitoring of bycatch and stock recovery, and breeding/stocking protocols. Measures designed to protect habitat used by Atlantic sturgeon while in spawning rivers is also protective of shortnose sturgeon.

5.13.3 Revised Use of TEDs in Trawl Fisheries

NMFS has also implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles in commercial shrimp trawl fisheries, which are also assumed to help Atlantic sturgeon. In particular, NMFS has required the use of TEDs in southeast U.S. shrimp trawls since 1989. It has been estimated that TEDs exclude 97% of the sea turtles caught in such trawls. These regulations have been refined over the years to ensure that TED effectiveness is maximized through more widespread use, and proper placement, installation, configuration (e.g., width of bar spacing), and floatation.

Atlantic sturgeon benefit from the use of devices designed to exclude other species from trawl nets, such as TEDs. A top-opening flynet TED was certified in the summer of 2007, but experiments are still ongoing to certify a bottom-opening TED. All of these changes may lead to greater conservation benefits for Atlantic sturgeon.

5.13.4 Smalltooth Sawfish

Public outreach efforts are also helping to educate the public on smalltooth sawfish status and proper handling techniques and helping to minimize interaction, injury, and mortality of encountered smalltooth sawfish. Information regarding the status of smalltooth sawfish and what the public can do to help the species is available on the websites of the Florida Museum of Natural History,[61] NMFS, and the Ocean Conservancy. These organizations and individuals also educate the public about sawfish status and conservation through regular presentations at various public meetings.

5.13.5 Corals

Education and outreach activities, as part of the NOAA Coral Reef Conservation Program, as well as through NMFS' ESA program, are ongoing through the Southeast Regional Office. NOAA's Restoration Center has also established a contract position in Puerto Rico to participate in grounding response and carry out restoration activities.

In addition, staghorn coral and elkhorn coral are commonly propagated in coral nurseries and outplanted back to the reef to supplement natural populations. Over 120,000 colonies of staghorn coral and over 7,000 colonies of elkhorn coral have been outplanted to over 140 Florida reefs over the last 10 years (NMFS, unpublished data) and have resulted in higher density at sites where population enhancement has occurred (Miller et al. 2016).

[61] http://www.flmnh.ufl.edu/fish/Sharks/Sawfish/SRT/srt.htm

5.13.6 Giant Manta Ray

Manta rays were included on Appendix II of Convention on International Trade in endangered Species of Wild Flora and Fauna (CITES) at the 16 Conference of the CITES Parties in March 2013, with the listing going into effect on September 14, 2014. Export of manta rays and manta ray products, such as gill plates, require Start CITES permits that ensure the products were legally acquired and that the Scientific Authority of the State of export has advised that such export will not be detrimental to the survival of that species (after taking into account factors such as its population status and trends, distribution, harvest, and other biological and ecological elements). Although this CITES protection was not considered to be an action that decreased the current listing status of the threatened giant manta ray (due to its uncertain effects at reducing the threats of foreign domestic overutilization and inadequate regulations, and unknown post-release mortality rates from bycatch in industrial fisheries), it may help address the threat of foreign overutilization for the gill plate trade by ensuring that international trade of this threatened species is sustainable. Regardless, because the United States does not have a significant (or potentially any) presence in the international gill plate trade, we have concluded that any restrictions on U.S. trade of the giant manta ray that are in addition to the CITES requirements are not necessary and advisable for the conservation of the species.

5.13.7 Johnson' seagrass

State and federal conservation measures exist to protect Johnson's seagrass and its habitat under an umbrella of management and conservation programs that address seagrasses in general (Kenworthy et al. 2006). These conservation measures must be continually monitored and assessed to determine if they will ensure the long-term protection of the species and the maintenance of environmental conditions suitable for its continued existence throughout its geographic distribution.

5.14 Impact of the Baseline on ESA-Listed Species

Collectively, the stressors described above have had, and likely will continue to have, lasting impacts on the ESA-listed species considered in this Opinion. Some of these stressors result in mortality or serious injury to individual animals, whereas others result in more indirect or nonlethal impacts. Assessing the aggregate impacts of these stressors on the species considered in this Opinion is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that many of the species in this Opinion are wide-ranging and subject to stressors in locations throughout and outside the action area.

We consider the best indicator of the aggregate impact of the *Environmental Baseline* on ESA-listed resources to be the status and trends of those species. As noted elsewhere in this Opinion, some of the species considered in this consultation are experiencing increases in population abundance, some are declining, and for others, their status remains unknown. Taken together, this indicates that the *Environmental Baseline* is impacting species in different ways. The species experiencing increasing population abundances are doing so despite the potential negative impacts of the *Environmental Baseline*. Therefore, while the *Environmental Baseline* may slow their recovery, recovery is not being prevented. For the species that may be declining in abundance, it is possible that the suite of conditions described in the *Environmental Baseline* is preventing their recovery. However, it is also possible that their populations are at such low levels that even when the species' primary threats are removed, the species may not be able to achieve recovery. At small population sizes, species may experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their limited population size to become a threat in and of itself. A thorough review of the status and trends of each ESA-listed species likely to be adversely affected by the proposed action is discussed in Section 4 of this Opinion.

6 EFFECTS OF THE ACTION

Section 3 of this Opinion discussed routes of effects expected to occur from activities covered under by Opinion, and identified those activities that are likely to cause adverse effects. We believe that the only activities covered under this Opinion that will result in adverse effects are those listed below, which will be analyzed further in this section:

- Hopper dredging will result in take of sea turtles (green, Kemp's ridley, and loggerhead) and sturgeon (Atlantic and shortnose).
- Cutterhead dredging will result in take of Atlantic and shortnose sturgeon in rivers
- Relocation trawling will result in take of sea turtles (green, Kemp's ridley, leatherback, and loggerhead), sturgeon (Atlantic and shortnose), and elasmobranchs (smalltooth sawfish and giant manta ray).
- Dredging will result in the removal of Johnson's seagrass.
- Beach nourishment, pipeline placement, and the associated relocation of ESA-listed corals will result in take of ESA-listed corals (elkhorn, staghorn, boulder star, lobed star, and mountainous star coral).

All activities must comply with the PDCs in Appendix B-Appendix H to be covered under this Opinion.

6.1 Effects to Mobile Species from Hopper Dredging, Cutterhead Dredging, and Relocation Trawling

To evaluate the effects and calculate take from hopper dredging, cutterhead dredging, and relocation trawling covered under this Opinion, we reviewed historical records of take of mobile species reported from these activities, as discussed in Section 3.1.1.4 for cutterhead dredging, Section 3.1.1.5 for hopper dredging, and Section 3.1.3 for relocation trawling.

6.1.1 Risk-based assessment

Timing of particular activities under SARBO will be determined using a risk-based assessment process, as described in Section 2.9.2.2 of this Opinion. Information gathered as part of the risk-based assessment process may reduce take by providing adaptability in when, where, and how dredging is completed so that the USACE and BOEM can continue to meet their dredging needs, while minimizing the risk of take of ESA-listed species. While these measures are intended to reduce the risk of take, the take estimated for cutterhead dredging, hopper dredging, and relocation trawling covered under this Opinion is still based on the historic reported take, as it is not possible to calculate the degree to which these efforts may in fact reduce take. Examples of types of considerations that the risk-based assessment will incorporate are below.

Equipment Choice
The 2020 SARBO allows options of equipment choice to complete work. Allowing the use of relocation trawling (discussed in Section 6.1.4,of this Opinion) is intended to reduce the lethal take expected from hopper dredging. Also the use of bed-leveling as an alternative to continued hopper dredging during the clean-up phase of dredging is expected to reduce lethal take, as the clean-up phase of hopper dredging can increase the risk of take by dredging in areas with uneven surfaces where the draghead may struggle to stay buried in the sediment.

Project timing
Seasonal variation and water temperature play an important role in the potential density of species in an area and the risk of entrainment, and USACE has indicated that it will consider seasonal variation when considering project timing, which may reduce the risk of take. For example, sea turtles use thermoregulation resulting in less movement when water temperatures drop, which may result in an increased risk of take during these times when they are less able to avoid dredging. This concern was discussed in a recent paper by BOEM (BOEM 2017), which discussed that lethal temperatures for sea turtles can be around 8 degrees C or below (Ogren and McVea Jr. 1982), but sublethal effects for low temperatures during overwintering (between 8 and 15 degrees C) can include immobilization, lethargy, lowered feeding rates, and starvation (Schwartz 1978) (Milton and Lutz 2003; Spotila et al. 1996). According to the NOAA (https://www.fisheries.noaa.gov/feature-story/cold-snaps-and-stunned-sea-turtles), cold stunning regularly occurs on the Atlantic coast in the Indian River Lagoon complex. In January 2018, more than 1,000 turtles, mainly juvenile greens, were rescued after becoming cold-stunned in St. Joseph Bay. The largest cold stunning event recorded in Florida occurred in 2010, with more than 4,500 sea turtles rescued within the state.

Turtles become "cold-stunned," or torpid and floating at the surface (Schwartz 1978), which may make them at greater risk to any activities at the surface or throughout the water column, such as fishing gear, vessel strikes, or hopper dredging operations with pumps engaged while dragheads are off the bottom to clear the lines. There have also been rare incidents suggesting that cold-stunned turtles may be taken by cutterhead dredges while they are lethargic or dying and unable to move away from the cutterhead, as discussed in Section 3.1.1.4.1 of this Opinion. Temperature will be considered as part of the risk assessment outlined in Appendix J, that dredging cease if water temperature drops below 11ºC to avoid the risk of encountering cold-stunned sea turtles that are at higher risk of take.

A study released by the USACE (Dickerson et al. 2007), also evaluated the risk of sea turtle entrainment and capture in relocation trawling during different times of the year. This study concluded that sea turtle entrainment is higher during the spring and fall within the action area and concluded this may be a result of a higher number of turtles in the area as they migrate up and down the coast. It was also noted that entrainment decreased during the summer (July-September) and was comparable to those observed during cold winter months, though the summer dredging sample size was significantly smaller. The study stated that the decrease in entrainment in the summer may be linked to an increase in activity of sea turtles during months were they are nesting and that they may spend more time moving through the water column, which would result in a lower risk of entrainment by the draghead operating at the sea floor.

Another example of a way the risk based assessment will be used is the determination to use hopper dredging in certain areas during warmer months than was covered under the 1997 SARBO. The USACE has stated that they plan to dredge some navigation channels during warmer months, outside of the North Atlantic right whale migration season, since navigation channels require the use of high speed survey vessels that are of greatest risk of a vessel strike to North Atlantic right whales. These locations do not have beach quality sand and therefore placement of the dredged material would not be used for beach nourishment projects during sea turtle nesting season. Therefore, the risk based assessment is used to weigh the risk of a vessel strike to a whale during months that these whales are present in the action area vs the risk of a sea turtle take by hopper dredging during months when sea turtles may be more abundant in the area, and if the material dredged would need to be placed when sea turtles are nesting. The decision of where and when to dredge will continue to be adjusted using the risk-based assessment process, which may include adjustments to this list of warmer month dredging locations. The navigation channels currently proposed to be dredged in warmer months include Brunswick Harbor, Savannah Harbor, Charleston Harbor, Wilmington Harbor Entrance/Inner Ocean Bar, Morehead City, and Manteo Entrance Channel, any additional locations will be evaluated using the risk-based assessment process.

Species habitat use
As more information about where and when other species use an area within the action area becomes available, this information will be incorporated into the risk assessment as well and may reduce the risk of take. For example, new information indicates that giant manta ray are being found in increased density in shallow waters off the east coast of Florida, particularly around St. Augustine. As we refine our understanding of their use of particular areas, this information will be shared with the USACE and BOEM as part of the annual review process and an evaluation of the best times and equipment types to use in projects, such as avoiding trawling during times and areas where the giant manta may be likely to be found in greater densities. Similarly, Atlantic sturgeon take reports indicate that more takes seem to occur more frequently at the mouths of certain rivers during winter months. This information is already being considered as part of the USACE internal risk assessment for continued maintenance dredging in these areas.

A BOEM study[62] also concluded that the risk of entrainment of sea turtles from hopper dredging in borrow sites in federal waters under their jurisdiction is historically lower than in navigation channels. The study stated that reported take was lower in borrow areas and concluded that this may be related to the fact that borrow sites in federal waters have a lower density of sea turtles. We support this assessment since sea turtle foraging and refuge habitat is not likely to occur in the sandy offshore areas selected for borrow sites and that these open water areas are easier for sea turtles to avoid construction than confined channels and that this information can be used as part of the risk assessment process when scheduling work. The BOEM study stated that between

[62] http://photoscience.maps.arcgis.com/apps/MapJournal/index.html?appid=6f8672c77ca94976aab12bd290e1fafd

1995 and 2017, only 25 sea turtles have been entrained while dredging OCS sand sources in federal waters. When compared to the 342 takes reported in the 2020 SARBO action area in Table 12 during nearly the same time period (1998-2018) that includes those reported in federal waters, it demonstrates that reported take in borrow areas in federal waters is lower than those reported in state waters. We believe that take in borrow sites in state waters are also lower than other nearshore projects for the same reasons.

Tracking information gathered
Compiling the species information gathered to use as part of the risk-based assessment is another challenge. However, new decision-making tools are being developed to aid in this process. For example, BOEM has developed such a tool that incorporates the available data on sea turtle use of areas in federal waters. Sea turtle tracking data is being uploaded into their analysis program, referred to as the Analyzing Sea Turtle Entrainment Risk (ASTER) decision support tool. Technical insight was gathered from both sea turtle and dredge expert communities who have a broad knowledge base and understanding of the relationship of dredging entrainment risk relative to sea turtle distribution and behavior, dredge operational parameters, and the implementation of existing mitigation measures. This information was used to inform the development of a standardized geographically and temporally based decision support tool for use by practitioners in the Atlantic and Gulf region to assess project-specific dredging entrainment risk within a common framework. BOEM's goal is that having more informed decisions may minimize impacts to sea turtle species while also decreasing dredging costs. We believe that the use of tools such as ASTER as part of the risk based assessment will aid in meeting BOEM's goal of improving decision making that will reduce take. We look forward to continued investments in sea turtle telemetry and the development of other similar tools that will help predict the likelihood of a species presence in a specific area at a set time of year.

For projects covered under this Opinion, PSO's will monitor the activities for evidence of take, according to the requirements of the PDCs (General PDC in Appendix B Section 3.1). When hopper dredging a particular section is complete, the PSO visually examines the draghead to determine if an ESA-listed species was impinged in the draghead. The ability to detect take at these locations is only partially effective and observed interactions likely provide only partial estimates of total mortality. NMFS believes that some proportion of ESA-listed species killed by hopper dredging go undetected. Mortalities are only observed and documented when body parts are impinged in the draghead, entrained in the inflow or overflow screening, or are viewed floating in the hopper. Body parts that are forced through the suction draghead and inflow screens by the suction-pump pressure and that do not float in the hopper are unlikely to be observed, since they will sink to the bottom of the hopper and not be detected by the overflow screening. The majority of sea turtles killed by hopper dredges are immediately crushed or dismembered by the heavy suction dragheads and/or by the force of the hopper dredges' powerful, high-velocity dredge pumps. Very few sea turtles (over the years, a fraction of a percent) survive entrainment in hopper dredges and those that do survive are typically smaller juveniles that are likely sucked through the pumps without being dismembered or badly injured. While those turtles that make it through a hopper dredge alive appear uninjured, they often die later of unknown internal injuries while in rehabilitation. The PSO PDCs in Appendix H Section 3.1 require that all sea turtles entrained alive be evaluated prior to release to determine if they can be safely rehabilitated or if their injuries are too severe. For the purposes of this Opinion, turtles entrained alive will be assumed to be lethal take until after they are evaluated by a sea turtle rehabilitation specialist, deemed healthy, and then released back into the wild.

6.1.2 Estimated Take from Hopper Dredging

As discussed in Section 3.1.1.5 of this Opinion, hopper dredging associated with the proposed action will result in take of sea turtles (green, Kemp's ridley, and loggerhead) and sturgeon (Atlantic and shortnose) from entrainment in the hopper dredge or impingement on the draghead. Based on the PSO PDCs in Appendix H Section 4, all observed take will be considered lethal even if the animal retrieved is alive and sent to rehabilitation until such time as it can be returned alive and healthy to the wild population.

To estimate take from hopper dredging covered under this Opinion, we reviewed the reported take of ESA-listed species from projects in the action area provided in Table 12 and compared it to the reported volume of material dredged (effort) by those projects to calculate a hopper dredging CPUE for each species[63]. We reviewed all reported lethal takes that occurred in the 21 years since the issuance of the 1997 SARBO (fiscal years 1998-2018, which is the last complete year of data available), as shown in Table 12. The number of reported observed takes from hopper dredging was gathered from information provided by the USACE from their public tracking website ODESS.[64] The reported total volume of material dredged per fiscal year in Table 12 was gathered from multiple sources provided by and/or verified by the USACE, but still may not precisely reflect the volumes dredged. At this time, it is the best available information and will be used for the analysis of take for this Opinion.

These historic records demonstrate that the number of take reported annually can fluctuate significantly. Some of the reasons for interannual variability include, but are not limited to, the location and type of projects, volume dredged, environmental conditions, changes in populations size and the density of animals in the area. A few examples of this variability are highlighted in the list below:

- Sturgeon spawn in rivers and therefore the range for this species extends further upriver than any other species in the action area. Within these rivers, areas of higher concentrations of sturgeon have been identified (i.e., seasonal aggregation areas) during certain times of year. In addition, recent dredging and relocation projects occurring at the mouth of spawning rivers have also resulted in an increased number of Atlantic sturgeon take in areas that did not previously have sturgeon take. It is unknown if the increased take was the result of an

[63] CPUE is calculated by comparing the number of observed reported takes (catch) to the cubic yards of material dredged (unit of effort).

[64] The numbers in Table 12 are calculated from a complete dataset provided by the USACE and not from the information available for download from ODESS by the general public. The version of ODESS available at the time of completion of this Opinion was a beta test and errors were discovered in the way that take was calculated on some of the available website pages (e.g., not all species with take are shown on different platforms reported take such as a table or pie chart, take reporting was inconsistent with some reports providing information by calendar year and others by fiscal year, and the volume of material reported dredged per year was incomplete and therefore inaccurate). NMFS is coordinating with the USACE on reporting requirements and quality control of the information reported and awaits the release of a new version of ODESS that resolves these errors, expected in the near future.

unusual event or if this is an area which sturgeon use consistently. Based on this information, projects occurring in any of these areas may result in a higher risk of take and based on the site conditions and season, the probability of encountering sturgeon may vary. The Sturgeon PDCs in Appendix E are designed to minimize risk by not allowing dredging to occur in known sturgeon aggregation areas at times of year when environmental conditions result in additional stress to sturgeon (referred to as the Sturgeon Summer Dredging Windows in the Sturgeon PDCs in Appendix E). The Sturgeon PDCs in Appendix E will continue to be updated as new information on sturgeon aggregation areas, high occupancy areas, and water quality data is acquired in order to minimize the risk of take.

- Areas near high density sea turtle nesting beach may also have an increased number of turtles in the area that are at increased risk of take. The 2020 SARBO considers dredging in channels near or adjacent to nesting beaches during warmer months than were allowed under the 1997 SARBO, which could result in a higher probability of encountering turtles during dredging and/or relocation trawling. Again, work occurring in these areas will be evaluated in the risk assessment process outlined in Section 2.9.2 of this Opinion to further minimize the risk of sea turtle take.
- Not all areas of potentially high density of a species are known. For example, information is still being actively gathered on where and when newly listed species like the giant manta ray occur within the action area. As information is gathered, it will be considered and minor modifications to protective measures may be made during the risk assessment process for future projects to further minimize the risk of take.

Variability in reported take from year to year may also be a result of environmental conditions. For example:

- Work may be scheduled during an unseasonably warmer or colder period that may change the risk of take. For example, sea turtles are known to be more vulnerable after a cold snap and may not actively avoid dredging leading to an increase in take, which will be discussed further in Section 6.1.1 of this Opinion.
- There may be changes in the species composition in an area that results in changes in density of ESA-listed species. For instance, a project in Dare County, North Carolina in 2017, resulted in the capture of 10 leatherback sea turtles in relocation trawling, which typically are rarely captured in trawling. This particular event was attributed to a high concentration of cannonball jellyfish in the area that the leatherback sea turtles had presumably followed to nearshore waters for foraging where trawling was occurring.
- The 2010 Louisiana Emergency Berm Project (outside of the action area) captured an unprecedented number of sea turtles during relocation trawling that may have been the result of the ongoing DWH oil spill event that resulted in an unprecedented amount of trawling and that the spill and recovery efforts may have modified turtle behavior and use of the area.
- Stochastic events such as major hurricanes may also change the density of species in the area as they respond to either the upcoming storm or changes in the environment following the storm.

We believe that estimating reported take that is expected to occur annually based on a 1 year estimated total would not adequately account for the annual variability in take from hopper dredging, cutterhead dredging, and relocation trawling (discussed below). For this reason, and based on our experience monitoring fisheries, we believe a 3-year time period is appropriate for meaningful monitoring, where possible. The triennial takes are set as 3-year running sums (total for any consecutive 3-year period) and not for static 3-year periods (i.e., fiscal year 2020-2022, 2021-2023, 2022-2024 and so on). This approach accounts for the inherent variability in take levels, while still allowing for an accurate assessment of how the proposed actions are performing versus our expectations. As noted by species in the analysis that follows, we diverge from this approach where very small take numbers are projected.

Due to the fluctuation in take reported, we compared the hopper dredge CPUE based on all 21 years of reported data since the issuance of the 1997 SARBO, as well as just the last 5 years (Table 36), to determine if take levels in recent years may better represent anticipated effects under 2020 SARBO. Table 36 provides the CPUE calculated for these periods of reported take relative to the reported volume of dredged material for the same time period. We then calculated the estimated number of take that may occur from hopper dredging annually under the 2020 SARBO based on the estimated average, minimum, and maximum volume of material that may be dredged annually. Dredging volumes are based on the information provided by the USACE and BOEM provided in Section 2.1 of this Opinion, which estimates that hopper dredging covered under this Opinion will account for an estimated annual dredge volume of between 10,899,182 and 27,386,533 cy, with an average of 17,599,004 cy dredged annually.

We compared the authorized take and reported observed annual take under the 1997 SARBO in Table 12 to see if, when, and how often take was exceeded and compared this information to the estimated take in Table 36 for 2020 SARBO and the following observations were made:

- The 1997 SARBO includes a take estimate for shortnose sturgeon; however, no take was reported for this species likely due to the limited amount of work that occurred in sturgeon spawning rivers where this species spend the majority of its life. However, the expanded action area for the 2020 SARBO includes more areas in sturgeon rivers and therefore consideration of these effects under the 2020 SARBO is warranted.
- The annual take observed under the 1997 SARBO was exceeded multiple times for green and Kemp's ridley sea turtles in the last 8 years. When this was compared to the nesting trends for green sea turtles (Figure 25) and Kemp's ridley sea turtles (Figure 26), it is clear that the number of reported nests per season, and therefore likely the population for these species, has also increased in recent years. Therefore, looking at take for sea turtles based on recent years reported seems warranted, to account for population trends.
- The 1997 SARBO also did not provide take for Atlantic sturgeon, which were listed after the completion of that Opinion. However, reported take in Table 12 includes all takes observed after the listing of the 5 DPSs in 2012. Therefore, reviewing take that has occurred more recently, as opposed to since 1997, will more accurately account for potential take of Atlantic sturgeon. The USACE previously considered the potential for these species to be jeopardized by the continued dredging operations occurring under the 1997 SARBO while in reinitiation of the 2020 SARBO under a 7(a)27(d) analysis under the ESA.

Table 36. Calculated Annual CPUE for Hopper Dredging[65]

	Dredge volume (cy)	Green sea turtle	Kemp's ridley sea turtle	Loggerhead sea turtle	Unknown sea turtle	Atlantic sturgeon	Unknown sturgeon
1997 SARBO Annual Take limit		7	7	35	0	0	0
Average annual take		3	3	9	0	2	0
Minimum annual take		0	0	3	0	0	1
Maximum annual take		10	8	18	4	14	0
Number of Years Take Exceeded (Table 12**)**		4	1	0	NA	5	NA
Total Reported (21 year total, 1997-2018)	176,817,454	53	54	183	5	39	3
Total Reported (5 year total, 2014-2018)	48,591,799	18	23	46	4	25	3
Additional projects (Table 12**)**		17	10	15		14	
Total		35	33	61	4	39	3
Calculated CPUE (5 year total, 2014-2018)		7.20286E-07	6.79127E-07	1.25536E-06	8.23184E-08	8.02605E-07	6.17388E-08
Total Annual Estimated Take (based on average dredged volume from Table 3**)**	17,599,004	12.68	11.95	22.09	1.45	14.13	1.09
Total Annual Estimated Take (based on maximum dredged volume from Table 3)	27,386,533	19.73	18.60	34.38	2.25	21.98	1.69

[65] CPUE calculated by dividing the number of reported takes per time period by the total volume dredged during the same time period. Estimated take calculated by multiplying the CPUE by the estimated dredge volumes provided in the second column. All take reported is lethal.

Based on a review of the reported take in Table 12, estimated take by the time periods shown in Table 36, and a review of the species population trends discussed in Section 4 of this Opinion, we estimate the level of take that will be observed annually from hopper dredging covered under this Opinion discussed by species, below.

While the average estimated volume dredged calculated in the Proposed Action is 17,599,004 cy we believe that the take associated with this estimated average annual hopper dredge volume (as calculated Table 36) results in an underestimate of the actual impacts of hopper dredging associated with the proposed action, for the following reasons:

1. Dredging will occur in areas and at times when certain ESA-listed species are expected to occur with higher concentrations than projects completed under the 1997 SARBO. As discussed in Section 2.5.2 of this Opinion and as part of the additional consultation history in Appendix K, the USACE plans to dredge some navigation channels during warmer months, outside of the North Atlantic right whale migration season, since navigation channels require the use of high speed survey vessels (which pose a higher threat of striking whales). The expanded ability under this Opinion to hopper dredge during summer months when sea turtle concentrations are expected to be higher will mean that there may be a higher chance of take, due to increased numbers of animals. The decision of where and when to conduct hopper dredging will continue to be adjusted using the risk-based assessment process outlined in Section 2.9.2 of this Opinion, which may include dredging of more locations during sea turtle nesting season. The navigation channels currently proposed to be dredged in warmer months (including sea turtle nesting season) include Brunswick Harbor, Savannah Harbor, Charleston Harbor, Wilmington Harbor Entrance/Inner Ocean Bar, Morehead City, and Manteo Entrance Channel. Other channels will be reviewed through the risk-based assessment process. The Kings Bay entrance may occur during any month, if deemed appropriate based on the initial risk-assessment analysis. Cape Canaveral also has not been dredged by hopper dredging since the late 1980's. Cape Canaveral is known to have a high density of green sea turtles and work in this area would result in a disproportionate number of take for this species. We therefore believe that the expansion of areas and times in which dredging may occur, will increase the number of reported takes, beyond those estimated in Table 36 by applying the 1997 SARBO CPUE to the annual average hopper dredge volume anticipated under the 2020 SARBO.
2. An increase in sea turtle populations is expected to be encountered over the lifetime of this Opinion. Green (NA and SA DPS), Kemp's ridley, and loggerhead sea turtle populations are increasing based on the information provided in the Status of the Species in Section 4.1.1 of this Opinion and discussed further in the Jeopardy Analysis in Section 8 of this Opinion.
3. An increase in the accuracy of take reports under the 2020 SARBO is expected, and therefore the amount of reported take, based on the new PSO guidance provided in Appendix H. This guidance is expected improved observations and therefore reported interactions with listed species. The PSO PDCs provide detailed guidance on how to detect and handle ESA-listed species that may be found in the action area. The USACE and BOEM provided their expertise on the operational issues encountered on dredging vessels and relocation trawlers and the USACE provided NMFS staff the opportunity to visit a hopper dredge, cutterhead dredge, and relocation trawler in operation. This provided a better understanding of the equipment operations and allowed NMFS to observe a PSO working on these projects to

understand their work challenges and limitations. We also coordinated with other NMFS regions to better describe the training necessary to complete work as a PSO (separate from this Opinion), which is expected to improve the quality of trained PSOs that will work on future projects covered under this Opinion. We believe that by improving the guidance provided to PSOs on how to observe take, and ensuring that hopper dredge intake and overflow areas are accessible to the PSO, the number of observed takes reported will increase on hopper dredging projects.

4. For sturgeon, projects covered under this Opinion will occur in more areas of sturgeon rivers than previously covered by the 1997 SARBO and may result in more interactions with both Atlantic and shortnose sturgeon.

Given the changes in the proposed action and action area from the 1997 SARBO, increasing population trends for sea turtle species that are susceptible to entrainment by hopper dredging, and improved accuracy of take monitoring and reporting under this Opinion, we believe that the 1997 SARBO CPUE applied to the estimated annual amount dredged under the 2020 SARBO results in an underestimate of take. However, the best data available to estimate future hopper take within the action area is from 1997 SARBO and previous projects in the action area. Therefore, we believe that it is most accurate to base the future estimated observed take on the hopper dredge CPUE during the last 5 years (Table 32), and then multiplying the CPUE by the maximum volume of hopper dredging estimated to occur annually under the 2020 SARBO (27,386,533 cy). Relying on this maximum estimated annual hopper dredge volume, solely for the purpose of estimating take from hopper dredging under this Opinion, is intended to ensure that the projected level of take is consistent with the range of dredging that is expected occur as a result of the proposed action, while also accounting for the expected increased rate of both the amount of take and accuracy of take reporting, for the reasons identified above.

We believe that the last 5 years of data represents the best available information to estimate take for these species, despite the limitations acknowledged above, because it most accurately captures recent abundance trends and is most reflective of USACE's expected activities in the future, as further explained below by species.

- Green, Kemp's ridley sea turtle, loggerhead sea turtle, and Atlantic sturgeon: As stated above, the last 5 years of data represents the best available information to estimate take for sea turtle species, because these populations have generally be increasing. We do not have similar trend data for Atlantic sturgeon, which were not listed until 2012. However, the increased captures of Atlantic sturgeon by both hopper dredging and relocation trawling, discussed later in this Opinion, indicate that using the most recent set of data from the last 5 years is warranted.
- Unknown sea turtles: Hopper dredging take of turtles can result in observed parts that cannot be identified to the species as has been reported in previous years and is expected to continue to occur under 2020 SARBO. Therefore, we assume that 2.25 turtles may be captured annually that cannot be identified from Table 36. For each sea turtle species we break out the expected take based on the percentage of lethal take expected for each (Green = 27.13%, Kemp's ridley = 25.58%, and Loggerhead = 47.29%). We then applied that percentage to the anticipated unidentified turtle take to estimate which species would comprise the unidentified takes. This analysis calculated annual take of up to 0.61 green, 0.58 Kemp's ridley, and 1.06

loggerhead sea turtles, as unidentified. For the Jeopardy analysis, we will assume that the 2.25 observed annual captures will equate to 3-year lethal take of up to 1.83 green, 1.73 Kemp's ridley, and 3.19 loggerhead sea turtles.

- Shortnose and Unknown sturgeon: Since dredging in sturgeon rivers was limited under the 1997 SARBO, we believe that it is appropriate to attribute the 1.69 annual takes calculated for unknown sturgeon in Table 36 to be considered shortnose sturgeon since this species is more likely to be encountered in sturgeon rivers once work begins in these areas. A limited number of projects have occurred in sturgeon rivers and did not report take of shortnose sturgeon. However, we still think it is likely that this species will be encountered since they co-exist with Atlantic sturgeon in rivers where Atlantic sturgeon take has been reported and shortnose sturgeon takes are reported in similar rivers in the northeast. We believe that hopper dredging will also continue to capture sturgeon parts that cannot be identified to the species as has been reported in previous years and is expected to continue under this Opinion. However, Atlantic sturgeon will be genetically tested to determine the DPS of the take and therefore unknown sturgeon will also need to be tested to determine if they are Atlantic sturgeon of a specific DPS. Hence, all sturgeon take will be identified using genetic testing and no unknown sturgeon will be considered for this Opinion.
- Hopper dredging in the U.S. Caribbean: The dredging volumes provided by USACE for anticipated hopper dredging projects in the U.S. Caribbean are included in the hopper dredge estimate for the proposed action. Species distributions in the U.S. Caribbean, and therefore hopper dredge take, may differ from the better known Atlantic waters from North Carolina – Florida. However, dredging projects off of the continental U.S. are the best available information available regarding levels of take of ESA-listed species by hopper dredge, as hopper dredging has not previously occurred in the U.S. Caribbean. Therefore, we consider the calculations in this section to incorporate the limited number of hopper dredging projects expected in the U.S. Caribbean.

While the final estimated take covered under this Opinion will be rounded to a whole number, rounding of take estimates will occur later in this analysis once all estimated take is calculated and combined in Section 6.1.5 of this Opinion. The final estimated take will also consider 3-year take average estimate discussed in Section 6.1 of this Opinion to account to variability, which will be calculated in combined take estimate in Section 6.1.5. As explained in Section 2.9.3.1 of this Opinion, USACE will report annual take to NMFS, but take levels will be tracked against the ITS in longer intervals (identified by species below), to better account for annual variability.

6.1.2.1 Ability to observe take in hopper dredging

It is not known how many sea turtles or sturgeon are taken by hopper dredging, but not observed. Historically, NMFS has estimated that only 50% of the take that occurs during hopper dredging is observed, and we have no new information to modify this estimate. Thus, our jeopardy analysis accounts for total take (observed take plus unobserved take), calculated in Table 37. We also assume that the 2.25 sea turtles will be observed captured that cannot be identified to the species.

Table 37. Estimated Annual Observed and Unobserved Hopper Dredging Take under this Opinion

	Hopper Dredging Observed	Hopper Dredging Unobserved	Hopper Dredging Total
Green sea turtle	19.73	19.73	39.45
Kemp's ridley sea turtle	18.60	18.60	37.20
Loggerhead sea turtle	34.38	34.38	68.76
Unknown sea turtle	2.25	2.25	4.51
Atlantic sturgeon	21.98	21.98	43.96
Shortnose Sturgeon	1.69	1.69	3.38

The estimated 50% of hopper dredging take assumed to be unobserved accounts for takes which may not be visually seen by a PSO due to pieces of the animal sinking to the bottom of the hopper, and not being caught by inflow/outflow screening, and also for the variability of observation effectiveness, such as if the ability to observe take is reduced based on the need to adjust screening on hopper dredges. For instance, the inflow and/or overflow screen size may need to be adjusted in certain situations to account for the amount of debris in the area being dredged or the composition of the material dredged. For example, dredging in Wilmington Harbor frequently results in the need to increase screening mesh size or remove the inflow screening to accommodate large woody debris or mud in the area, which clog the inflow screens rendering them ineffective to monitor for take. In limited instances, MEC/UXO screens are also needed to be placed on the draghead to exclude materials in the area such as rocks encountered as a borrow area is dredged to the rock below the sand intended to be removed. In these instances, MEC/UXO screening may be used as a safety precaution to protect the dredge from damage resulting from suctioning up damaging materials or potentially even unexploded munitions. In all of these scenarios, the ability to observe potential take is decreased. If inflow screens are increased in size, larger pieces of species may pass through undetected. If inflow screens are removed entirely, take of species cannot be observed in the inflow and observation of take is limited to only overflow screening. Overflow screening varies by hopper dredge vessel and may not as effectively capture species parts to monitor take. Overflow also only captures parts that float to the surface of the hopper and are entrained as water collected in the hopper is overflowed to make room for more dredged sediment to be held in the hopper for transport to the disposal location. While it is believed that species parts generally float, we assume that at least a portion are buried in the sediment contained in the hopper and are undetected. If MEC/UXO screening is placed on the draghead of the hopper, the screening is so small that it is assumed that take is not entrained through the draghead into the hopper and is therefore not observed. In limited instances, species have been reported impinged on the MEC/UXO screening, but it is assumed that most take is excluded and unobserved.

The USACE estimated that the need for MEC/UXO screening will be limited and all instances where this screening is used on a project covered under this Opinion will be reviewed by NMFS under the Supersede procedures outlined in Section 2.9.5 of this Opinion. This secondary review of projects requiring MEC/UXO screening will be considered on a case-by-case basis to ensure that the use of the screening is truly limited and that unobserved take is not expected to rise above the estimated 50%. Other considerations that will be reviewed include the likelihood of take based on the site conditions, likelihood of take based on expected species presence depending on the location and time of year, and length of time that MEC/UXO screening will be needed. In addition, NMFS will be notified when inflow screening is increased or removed and these scenarios will be considered as part of the annual review process.

Based on information provided by the USACE on how and when these scenarios occur and a review of previous projects that required these changes, we believe that the 50% unobserved take estimate will continue to account for even these instances where the ability to monitor for take is decreased.

6.1.3 Effects of Cutterhead Dredging

As discussed in Section 3.1.1.4.2 of this Opinion, we believe that cutterhead dredging in sturgeon rivers may result in sturgeon take. We anticipate no other ESA-listed species may be incidentally taken by cutterhead dredging, for the reasons discussed in Section 3.1.1.4 of this Opinion. Because of the specific requirements establish for monitoring take by cutterhead dredging in the Sturgeon PDCs (e.g., twice daily monitoring of the disposal pond), we do not anticipate sturgeon takes will go unobserved.

We anticipate sturgeon takes may occur when water quality in certain river reaches is poor. We identified these areas and times with the letters “B” or “C” in Table 56 in the Sturgeon PDCs in Appendix E. In the areas and times identified as “B” in Table 56 in the Sturgeon PDCs in Appendix E, water quality is poor, the section of the river has been surveyed, and no seasonal aggregation areas were identified. However, some of the rivers with poor water quality still support areas with higher densities of sturgeon. Unlike the discrete seasonal aggregation areas that appear to be associated with a physical feature, these “B” river reaches have relatively high sturgeon densities over a larger area without association to an obvious physical feature. These reaches are also used for longer durations than seasonal aggregations areas. Instead, these areas of high occupancy are likely preferentially selected by the animals because they provide beneficial environmental (e.g., appropriate temperature and DO concentrations) or resource (e.g., prey) conditions.

For example, telemetry data obtained in the Cape Fear River, North Carolina, indicate high occupancy levels of sturgeon along a relatively long stretch river (River Mile [RM] 18.75-37.5; River Kilometer [RKM] 30-60), from May-October. Based on the available bathymetry of this area, the area they are using does not appear to have a unique/specific physical feature (e.g., no deep holes), unlike aggregation areas observed in other rivers (e.g., Savannah River, Georgia/South Carolina). Because of the size and relative homogeneity of the river bottom in these areas of high occupancy, we anticipate that water quality changes caused by dredge-related activities would not cause adverse effects by forcing sturgeon from areas of refuge into surrounding environments that are unable to support their physiological needs. However, even though these animals may not ultimately be forced from areas of refuge, they may still be at greater risk of injury from equipment because they occur in relatively higher densities, and could suffer from reduced fitness because of low DO concentrations. In these circumstances, we believe there is an elevated risk of incidentally taking those animals by cutterhead dredging. As a result, the Sturgeon PDCs do not cover the use of hopper dredging in high occupancy areas (e.g., those rivers/times identified as “B” in Appendix E), to minimize the risk of take.

In the areas and times identified as “C” in Table 56 in the Sturgeon PDCs in Appendix E, water quality is also poor, the section of the river has been surveyed, and seasonal aggregation areas have been identified. For example, the known seasonal aggregation areas in the Savannah and Cooper Rivers are located association with specific physical features (i.e., deep holes or river bens) that are upriver of most of the proposed dredging areas covered under this Opinion. There are no areas of high occupancy identified in these rivers. Based on available information, we expect that most of the sturgeon in these rivers will be in the aggregation areas during periods when water quality is stressful for sturgeon. Because the individuals tend to stay in these relatively discrete locations, and the Sturgeon PDCs prohibit cutterhead dredging in the seasonal aggregation areas, we believe the risk of take by cutterhead dredging outside of the aggregation areas will be lower. Nonetheless, we anticipate some incidental take associated with cutterhead dredging may occur, due to the increased presence of sturgeon and relatively lower water quality. The following sections describe our approach for estimating incidental take.

Cutterhead dredging in sturgeon rivers in the action area has not previously been monitored. However, we do have reports of takes of sturgeon from cutterhead dredging similar to what is contemplated in the proposed action. As stated in Section 3.1.1.4.2, NMFS Greater Atlantic Region reports that 5 cutterhead dredge takes of shortnose sturgeon in the James and Delaware rivers, described in Reine et al. (2014), occurred in known overwintering aggregation areas, where “shortnose sturgeon rest on the bottom and exhibit little movement and may be slow to respond to stimuli such as an oncoming dredge” (NMFS 2018c). We believe sturgeon in the Southeast exhibit similar “hunkering” behavior in certain rivers during summer months when water temperatures are high and DO concentrations are low, as discussed in Section 3.1.1.2. We believe dredging during times when water quality is poor and sturgeon are stressed, that they are at an increased risk of entrainment in cutterhead dredging, similar to what occurred in the James and Delaware rivers. To estimate the potential impacts from cutterhead dredging, we reviewed the information available from monitoring these dredge types in James and Delaware rivers. Given the limited available information regarding sturgeon takes associated with cutterhead dredging, and the needs of our analysis, this represents the best available information from which to estimate sturgeon take as a result of cutterhead dredging in sturgeon rivers. Ultimately, we created Southeast-specific CPUEs for sturgeon, based on the information available from the Greater Atlantic Region. To establish Southeast-specific CPUEs, we had to be able to scale the CPUE estimated for the Northeast. Because population estimates of shortnose sturgeon are only available for Delaware River (a necessary requirement for scaling in our analysis), we did not consider the James River CPUE in our analysis.

NMFS (NMFS 2018b), reported 2 shortnose sturgeon takes during cutterhead dredging in the Delaware River. Using the information on sturgeon cutterhead takes from outside the action area and the information on the dredging likely to occur in sturgeon rivers in the action area, we estimated the potential take likely to occur as a result of the proposed action. Specifically, our approach included the following steps:

1) Determine which cutterhead projects are likely to occur in areas where sturgeon were at increased risk of interaction due to poor water quality (i.e., "B" and "C") sections of rivers.
2) Estimate CPUE from observed cutterhead dredging take in the Delaware River.
3) Scale CPUEs and estimate shortnose and Atlantic sturgeon take for proposed dredging based on scaled CPUEs and projected effort (e.g., cubic yards of material dredged) from Step 1, taking into account conservation benefits of Surgeon PDCs.

Step 1: Determine cutterhead projects occurring during the defined times and areas of "B" and "C" sections of rivers
Based on information provided by the USACE regarding anticipated dredging activities in the action area from FY 2018-2022, we determined what cutterhead dredging is likely to occur in rivers with "B" or "C" dredge window restrictions (denoted in Table 56) when animals are at increased risk of entrainment (i.e., months with high water temperatures and low DO concentrations). This exercise identified the Cape Fear River, North Carolina; Cooper River, South Carolina; and Savannah River, South Carolina/Georgia as areas where dredging would occur during sensitive times. Specifically, during times of higher water temperatures and low DO concentrations we anticipate:

- 680,556 cubic yards of material would be dredged from the Cooper River
- 356,250 cubic yards of material would be dredged from the Cape Fear River
- 2,712,329 cubic yards of material would be dredged from the Savannah River

Step 2: Estimate CPUE for shortnose sturgeon in Delaware River
The likelihood of interactions with cutterhead gear is a function of the amount of material dredged and the sturgeon population size where dredging is occurring, along with other factors. NMFS Greater Atlantic Region (NMFS 2018b) reported that 2 shortnose sturgeon were entrained in the Delaware River following the completion of dredging a total 509,946 cy of material. Based on this information we calculated a mean CPUE of 3.92×10^{-6} sturgeon takes per cubic yard of material dredged.[66] The most recent estimates for shortnose sturgeon populations in the Delaware River as reported in NMFS (NMFS 2010b) as 12,047 adult shortnose sturgeon (95% CI = 10,757 – 13,580) (ERC 2006). We assume that the probability of take should scale with population size and total effort; as such, we scaled from the Delaware River CPUE to account for differences in sturgeon population sizes in the action area in Step 3.

Step 3: Scale CPUEs and estimate take of shortnose and Atlantic sturgeon for proposed cutterhead dredging based on scaled CPUEs and projected effort (e.g., cubic yards of material dredged) from Step 1, taking into account for conservation benefits of Sturgeon PDCs.
As noted previously, we anticipate cutterhead takes of sturgeon could occur in the Savannah, Cooper, and Cape Fear rivers. The analysis below relies on the CPUE for shortnose sturgeon take calculated in Step 2 for cutterhead dredging on the Delaware River to calculate a scaled CPUE for the estimated shortnose sturgeon population in the Savannah, Cooper, and Cape Fear rivers (as applicable). The analysis also estimates take of Atlantic sturgeon in each river, by relying on the scaled CPUE. Estimates of CPUE for cutterhead dredging and Atlantic sturgeon

[66] 2 sturgeon entrainments ÷ 509,946 cy dredged material = 0.00000392 sturgeon/1 cy dredged material.

are not currently available either from the Southeast or Northeast. Given the similarities in life history between shortnose and Atlantic sturgeon, and because both species appear to display the same behaviors during periods of environmental stress, we believe using the estimated CPUE for shortnose sturgeon is appropriate to apply to Atlantic sturgeon.

Cooper River – Shortnose sturgeon: The most recent adult shortnose sturgeon population estimates for the Cooper River are reported in NMFS Biological Opinion for the Re-licensing of the South Carolina Public Service Authority Hydroelectric Project (FERC #199-205), SERO-2018-00325, (NMFS 2020a) as approximately 229 spawning adults. This estimate is likely conservative because it may omit non-spawning adults. By comparing this population estimate to that of Delaware River, we estimated the Cooper River population is approximately 1.78% the size of the Delaware River population.[67] We used this ratio to update our CPUE accordingly.[68]

We coordinated with South Carolina Department of Natural Resources to determine what proportion of sturgeon were likely to "stray" outside our defined seasonal aggregation areas. We applied the estimated "straying rate" to our estimated population sizes in these rivers, generating an estimate of the number of animals that are expected to be outside of the seasonal aggregation areas and therefore still vulnerable to cutterhead dredging after PDC implementation.

The information provided by the South Carolina Department of Natural Resources indicated that from 2016-2018, 0% of telemetered shortnose sturgeon were detected outside the defined time and area of the seasonal aggregation in the Cooper River. This suggests the seasonal aggregation area defined in the Sturgeon PDCs will be highly effective in protect shortnose sturgeon in the Cooper River. Thus, we anticipate cutterhead dredging in the Cooper River is extremely unlikely to adversely affect shortnose sturgeon.

Cooper River – Atlantic sturgeon: Using a similar approach, we compared the shortnose sturgeon and Atlantic sturgeon populations in the Cooper River to estimate potential takes of Atlantic sturgeon from cutterhead dredging. No adult Atlantic sturgeon population estimates have been performed for the Cooper River; however, the Atlantic sturgeon final listing rule estimates no more than 300 spawning adults are likely in the Cooper River (77 FR 5914; February 6, 2012). Thus, we anticipate the populations of adult Atlantic and shortnose sturgeon in the Cooper River may be similar in size. Using the shortnose sturgeon CPUE for the Cooper River and the South Carolina Department of Natural Resources straying rate of 24.94% for Atlantic sturgeon in the river, we estimated cutterhead takes of 0.012 Atlantic sturgeon every year in the Cooper River, or 0.035 over consecutive 3-year periods.[69] Because of the nature of cutterhead dredging, we anticipate these takes will be lethal.

Cape Fear River – Shortnose sturgeon: Shortnose sturgeon are considered extremely rare in North Carolina. Only occasional sightings of shortnose sturgeon are reported in the Cape Fear

[67] 229 spawning adult shortnose sturgeon in the Cooper River ÷12,047 adult shortnose sturgeon in the Delaware River = 0.017896218
[68] Delaware River CPUE (0.00000392 sturgeon/1 cy dredged material) x 1.78% Cooper River population relative to Delaware River population = Cooper River CPUE of 0.00000007 sturgeon/1 cy dredged material.
[69] 680,556 cy dredged material annually on average in Cooper River during seasonal aggregation period x Cooper River-specific CPUE 0.00000007 sturgeon/1 cy dredged material x 24.94% straying rate x 3 years = 0.035

River and those individuals may be transients from the Winyah Bay system in South Carolina. For this reason, we anticipate effects from cutterhead dredging to shortnose sturgeon in the Cape Fear River will be extremely unlikely.

Cape Fear River – Atlantic sturgeon: There are no estimates for Atlantic sturgeon populations in the Cape Fear River. However, the 2007 Atlantic sturgeon status review (Atlantic Sturgeon Status Review Team 2007) reports a total of 75-100 Atlantic sturgeon were captured during each of 2 separate gillnet surveys targeting Atlantic sturgeon, conducted during the 1990s. The Atlantic sturgeon final listing rule also suggests that no more than 300 spawning adults are likely in the Cape Fear River (77 FR 5914; February 6, 2012). This information suggests the populations of Atlantic sturgeon in the Cooper and Cape Fear rivers are comparable. Based on information provided by the USACE, we estimated the average amount of material dredged from the Cape Fear River annually during periods of poor water quality is approximately 356,250 cubic yards, or approximately half of what is projected to be removed from the Cooper River. No seasonal aggregation areas were identified in the Cape Fear River, so no seasonal protections were established there and straying rates are not applicable. Based on this information we qualitatively estimate that interactions between cutterhead dredges and Atlantic sturgeon in the Cape Fear River are likely similar to those in the Cooper River (i.e., 0.012 Atlantic sturgeon every year, or 0.035 over consecutive 3-year periods). Because of the nature of cutterhead dredging, we anticipate these takes will be lethal.

Savannah River – Shortnose sturgeon: We estimated the Savannah River-specific CPUE for shortnose sturgeon following the same steps laid out for the Cooper River. The most recent adult shortnose sturgeon population estimates for the Savannah River come from Bahr and Peterson (2017). They reported the adult population varied annually from 1,865 (784–4,694) individuals in 2013, 1,564 (1,005–2,513) in 2014, and 940 (535–1,753) in 2015. We acted conservatively and selected the mean estimate of 1,865; using this value will estimate the highest take rate. Following the same approach of comparing the Savannah River population estimate to that of the Delaware River population, we determined the former is approximately 14.5% of the Delaware River population; we used this ratio to scale our CPUE accordingly.[70,71]

Similar information on straying rate, provided by the South Carolina Department of Natural Resources for the Savannah River, indicated that from 2016-2018, 28.47% of telemetered shortnose sturgeon were detected outside the defined time and area of the seasonal aggregation. Since cutterhead dredging will only be allowed outside the aggregation areas, we refined our CPUE-based take estimates to account for only the portion of individuals likely to occur outside the defined time and area of the seasonal aggregation. Multiplying this re-scaled CPUE by the projected effort during the seasonal dredge window, we estimated cutterhead takes of 0.45 shortnose sturgeon every year, or 1.35 over consecutive 3-year periods.[72] Because of the nature of cutterhead dredging, we anticipate these takes will be lethal.

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[70] 1,865 adult shortnose sturgeon in the Savannah River ÷12,047 adult shortnose sturgeon in the Delaware River = 0.145748671

[71] Delaware River CPUE (0.00000392 sturgeon/1 cy dredged material) x 14.57% Savannah River population relative to Delaware River population = Savannah River CPUE of 0.00000057sturgeon/1 cy dredged material.

[72] 2,712,329 cy dredged material in Savannah River during seasonal aggregation period x Savannah River-specific CPUE 0.00000057 sturgeon/1 cy dredged material x 28.47% straying rate x 3 years = 1.35 shortnose sturgeon.

Savannah River – Atlantic sturgeon: As with our other rivers, estimates of adult populations of Atlantic sturgeon in the Savannah River are not available. The Atlantic sturgeon final listing rule estimates no more than 300 spawning adults are likely in the Savannah River (77 FR 5914; February 6, 2012). Thus, we anticipate the population of adult Atlantic sturgeon (300) in the Savannah River is approximately 16% of the adult shortnose sturgeon population in the river. The South Carolina Department of Natural Resources information on "straying" indicates 23.97% of telemetered Atlantic sturgeon were detected outside the defined time and area of the seasonal aggregation, less than shortnose sturgeon. Using the shortnose sturgeon CPUE for the Savannah River, but scaled to the likely smaller adult Atlantic sturgeon population in the river, and the South Carolina Department of Natural Resources straying rate, we estimated cutterhead takes of 0.06 Atlantic sturgeon every year in the Savannah River, or 0.18 over consecutive 3-year periods.[73] Because of the nature of cutterhead dredging, we anticipate these takes will be lethal.

Table 38. Estimated Shortnose and Atlantic Sturgeon Takes in Cutterhead Dredging, over consecutive 1-year and 3-year periods

River (Atlantic Sturgeon DPS)	Shortnose sturgeon – 1 Year	Shortnose sturgeon – 3 Years	Atlantic sturgeon – 1 Year	Atlantic sturgeon – 3 Years
Cooper River (Carolina DPS)	0	0	0.012	0.035
Cape Fear River (Carolina DPS)	0	0	0.012	0.035
Savannah River (SA DPS)	0.45	1.35	0.06	0.18

6.1.4 Effects of Relocation Trawling

As discussed in Section 3.1.3 of this Opinion, relocation trawling is covered under this Opinion. While relocation trawling is intended to reduce lethal take from hopper dredging, the process of relocating ESA-listed species is, in and of itself, a form of take under the ESA. Relocation trawling is monitored by PSOs trained to handle these species to minimize the risk of harm to them during relocation. The PSO PDCs in Appendix H provide handling and reporting guidance for ESA-listed species captured during relocation trawling. Additional PDCs regarding the time and locations where relocation trawling can occur are provided in the General PDCs in Section 3.5 of Appendix B, which limit tow times to 42 minutes to minimize the risk of adverse effects on ESA-listed species, primarily mortality of sea turtles due to forced submergence (National Research Council 1990a) (Epperly et al. 2002). Relocation trawling in the Caribbean is not covered under this Opinion.

Based on the historic records of relocation trawling in the action area (Table 13), 34 projects have used relocation trawling. This information was then used to determine that relocation trawling projects had an average of 929 tows per project and a maximum of 5,001 tows in a single project (Table 39). The USACE provided a list of projects anticipated in the next 5 years, which is representative of typical activities under the proposed action, of which an average of 31 projects are scheduled to be completed by hopper dredging and may use relocation trawling

[73] 2,712,329 cy dredged material in Savannah River during seasonal aggregation period x shortnose sturgeon Savannah River-specific CPUE 0.00000057 sturgeon/1 cy dredged material x 16% Atlantic sturgeon population relative to shortnose x 23.97% straying rate x 3 years = 0.18 Atlantic sturgeon.

annually. For the analysis in this section, we will accordingly assume that 31 projects a year will implement relocation trawling that will be covered under this Opinion. The average and maximum trawling effort that may occur each fiscal year is calculated by multiplying the average (929) and maximum (5,001) tows reported per project in by an estimated 31 projects that may need relocation trawling annually under this Opinion. Based on this data, we calculated an average of 28,779 (929 x 31 = 28,779) tows may be completed annually under this Opinion with a maximum of 155,031 tows (5,001 x 31 = 155,031) each fiscal year. This information is used to calculate the number of ESA-listed species that may be captured using relocation trawling under this Opinion (Table 41). For sea turtles and sturgeon, the estimated reported take is included in Table 40. Though there were no reported captures of giant manta ray and smalltooth sawfish, the potential for relocation trawling captures under this Opinion is possible as discussed for each species in this section.

Table 39. Capture Relocation Trawling Calculations

Location	Total Trawling Tows	Atlantic Sturgeon	Green Sea Turtle	Kemp's Ridley Sea Turtles	Leatherback Sea Turtles	Loggerhead Sea Turtle	Total Turtles	Total
All Reports (fiscal years 1997-2019)	31,595	297	53	91	25	358	527	824
Average per Project	929	10	2	3	1	11		
Minimum per Project	22	0	0	0	0	0		
Maximum per Project	5,001	79	29	19	10	90		
Percent sturgeon and turtles relocated	22	100%	10.06%	17.27%	4.74%	67.93%		
Percent of total captures	5,001	36%	6.43%	11.04%	3.03%	43.45%	56.55%	100%

6.1.4.1 Effects to Sea Turtles from Relocation Trawling

As turtles rest, forage, or swim on or near the bottom, trawls pulled across the bottom at 1.5 to 3 knots can sweep over them. Trawl nets typically have an overhanging head rope to prevent fish and shrimp from jumping over the mouth of the net when they are hit by the tickler chain or footrope. This overhang also prevents most sea turtles from escaping the trawl by heading for the surface. In clear water (e.g., during TED testing when videoed), some turtles may be able to determine an evasive tactic and swim forward, as documented in video. Also, turtles released at the mouth of the net during testing will behave differently than naturally encountered turtles, which could be resting/sleeping on the bottom and would quickly fall back into the net. Because of the trawl's greater speed or the sea turtles' eventually tiring, the sea turtles gradually drop back toward the rear of the net where they fall into the cod end and are caught. Captured sea turtles upon retrieval of trawl gear may be found dead, comatose, or alive and otherwise seemingly healthy, depending on the extent of forced submergence effects. Due to the 42-minute tow time limit of relocation trawling under this Opinion, however, we expect most sea turtles to avoid drowning or other serious consequences of forced submergence. The relocation trawling time limit has remained 42 minutes for all Section 7 dredging consultations, dating back at least to the 1997 SARBO. According to a study conducted by the USACE ERDC (Dickerson et al. 2007) on relocation trawling, it stated that the average tow time was 35 minutes, which the USACE has stated is to ensure that the maximum allowed time is not exceeded. The dredging consultation covering maintenance dredging and material placement in the Gulf of Mexico (NMFS 2007b), described the history of the 42-minute tow time as follows: "The National Research Council (1990a) report "Decline of the Sea Turtles: Causes and Prevention" suggested that limiting tow durations to 40 minutes in summer and 60 minutes in winter would yield sea turtle survival rates that approximate those required for the approval of new TED designs, i.e., 97%." We continue to believe that the 42-minute trawling limit will result in nonlethal (as opposed to lethal) take in the vast majority of instances, as discussed further below.

6.1.4.1.1 Effects of Forced Submergence

Sea turtles are air-breathing reptiles; thus they need to be able to reach the surface to breathe. Although they are able to conduct lengthy dives, most voluntary dives by sea turtles appear to be an aerobic metabolic process, showing little if any increases in blood lactate and only minor changes in acid-base status (i.e., pH level of the blood). In contrast, sea turtles that are stressed as a result of being forcibly submerged in trawls maintain a high level of oxygen consumption, which can rapidly consume their oxygen stores and can result in large, potentially harmful internal changes. Those changes include a substantial increase in blood carbon dioxide, increases in epinephrine and other hormones associated with stress, and severe metabolic acidosis caused by high lactic acid levels. The rapid oxygen consumption triggers anaerobic glycolysis, which can significantly alter their acid-base balance (i.e., pH level of the blood), sometimes to lethal levels (Lutcavage and Lutz 1997). Recovery to pre-submergence lactate levels can take several hours (Stabenau and Vietti 2003) to as many as 20 hours (Lutz and Dunbar-Cooper 1987). The rate of acid-base stabilization depends on the physiological condition of the turtle (e.g., overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g., water temperature, wave action, etc.), and the nature of any injuries sustained at the time of submergence (National Research Council 1990a).

It is likely that the rapidity and extent of the physiological changes that occur during forced submergence are functions of the intensity of struggling underwater, as well as submergence time (Lutcavage and Lutz 1997). Other factors potentially influencing the severity of effects from forced submergence include the size, activity level, and condition of the sea turtle; the ambient water temperature; and if multiple forced submergences have recently occurred. Disease factors and hormonal status may also influence survival during forced submergence. Because thyroid hormones appear to have a role in setting metabolic rate, they too may play a role in increasing or reducing the survival rate of a captured sea turtle (Lutcavage and Lutz 1997).

In the worst scenario, sea turtles drown from being forcibly submerged. Such drowning may be either "wet" or "dry." With wet drowning, water enters the lungs, causing damage to the organs and/or causing asphyxiation, leading to death. In the case of dry drowning, a reflex spasm seals the lungs from both air and water. Before drowning occurs, sea turtles may become comatose or unconscious, generally unresponsive, and with a drastically suppressed heart and respiration rate–indicative of at least a physiological injury.

While observers have documented the majority of sea turtles captured by relocation trawlers as alive and otherwise seemingly healthy, there have been instances of sea turtle mortalities listed below:

- In June 2010, several relocation trawlers working on an emergency dredging project in Louisiana related to the DWH oil spill event captured 194 sea turtles in just 15 days. There were 3 lethal takes, as well as one unresponsive sea turtle capture (later rehabilitated), that resulted in a 2.06% observed mortality rate, though this project could be considered anomalous due to the ongoing and potentially related offshore DWH oil spill event.
- Of all of the reported relocation trawling in Table 13, only 2 sea turtles were reported as lethal captures, resulting in a 0.38% observed mortality rate of sea turtles during relocation trawling.
- A USACE study (Dickerson et al. 2007) evaluating the effectiveness of relocation trawling stated that of the 1,239 sea turtles relocated between 1980-2006 throughout the action area, the northeast Atlantic coast, and the Gulf of Mexico, only 4 lethal takes (1 green, 2 loggerheads, 1 leatherback) resulted from injuries sustained during trawling capture and not dredging. The green turtle and one of the loggerhead deaths occurred as a result of injuries sustained when they became entangled in the net webbing. One loggerhead appeared to have drowned. The leatherback death occurred when the trawl net became caught on a sunken boat and the net remained submerged for several hours before the trawl rigging could be released. That would equate to 0.32% observed mortality in all reported relocation trawling considered in the study.

We determined that the estimated lethal captures during the DWH spill response is not appropriate to use as a comparison to normal trawling operations, due to the extenuating circumstances of a large oil spill. We believe that the average of the lethal take information that we reviewed in is the most accurate estimate of mortality associated with the proposed action. The rate of mortality observed in Table 13 (.38%) is based on actions that have previously occurred within the action area and is also similar to the observed mortality in the USACE study that evaluated a larger data set of relocation trawling including areas outside the action area of this Opinion. Therefore, we will assume that 0.38% of relocation trawling captures may result in lethal take. This seems to be an appropriate mortality estimate based on the available data reporting that most projects had no reported lethal take associated with relocation trawling, and that the 2020 SARBO provides additional PSO handling guidance in Section 5 of Appendix H, which is designed to further reduce lethal take and includes information on how to handle unresponsive sea turtles including resuscitation per the sea turtle conservation regulations (50 CFR 223.206(d)(1)(B)).

6.1.4.1.2 Effects to Gravid Female Sea Turtles

In addition to respiratory and metabolic stress, sea turtles can also exhibit dynamic hormonal/endocrine responses to stress (Jessop et al. 2002). Of particular concerns for relocation trawling, are the effects to sea turtles during important life stages such as gravid (i.e., pregnant, carrying eggs) female sea turtles approaching nesting beaches during the summer months that may be relocated as part of a project covered under this Opinion. Studies showed that female green turtles during the breeding season exhibited a limited adrenocortical stress response when exposed to ecological stressors and when captured and restrained. Other information indicates stress and exertion from forced submergence on gravid females could result in nonlethal reproductive loss. Forced submergence from relocation trawling can cause nonlethal reproductive loss (i.e., loss of a clutch of sea turtle eggs), which could result in partial or complete disruption of nesting. A partial loss can be treated as the loss of a nesting female sea turtle's single clutch (without effect on development and the ability to lay eggs in subsequent clutches for the year), while a complete disruption would be considered as the loss of all further nesting for that year. Beyond specific reproductive effects are the behavioral alterations and energetic losses resulting from displacement as turtles are relocated from the dredging area and potentially away from desired foraging areas and the nesting beach itself. In contrast, some sea turtles have been observed to exhibit continuation of nesting despite significant events, such as severe injury.

Some sea turtles captured during relocation trawling operations may return to the dredge site and could be subsequently recaptured. Captured sea turtles are typically relocated a significant distance from the dredge area (e.g., historically 2-5 miles), and past documented return times of recaptured sea turtles has been on the order of days. We believe that there is sufficient time to allow a sea turtle to recover fully from the effects of a previous capture. The 2020 SARBO PDCs describe how far sea turtles will be relocated depending on species (PSO PDCs, Appendix H), and if followed, we do not expect any significant forced submergence effects stemming from repeat captures. However, we will consider the effects to gravid sea turtles that may be more susceptible to stress and harm from relocation trawling.

For this Opinion, we will consider post-interaction mortality due to stress and exertion on nesting females, as well as the potential loss of clutch, due to the effects of relocation trawling in the warmer, summer months. We adopted a procedural directive (NMFS 2017e) to determine post-interaction mortality of sea turtles captured in trawl, net, and pot/trap fisheries. This directive specifically excludes directed trawl captures authorized under NMFS Section 10(a)(1)(A) permits or Biological Opinions (e.g., relocation trawling), based on required protective measures intended to maximize survival under these circumstances. However, the directive did not consider potential discrete effects on gravid female sea turtles. Subsequent examination and expert opinion on the issue indicate that there should be additional consideration of the effects of relocation trawling occurring in the summer months, which may affect nesting sea turtles. Capture in relocation trawls can subject gravid female sea turtles to physiological and traumatic effects such as stress, over-exertion, and internal injury (Harms et al. 2003; Phillips et al. 2015; Snoddy et al. 2009; Snoddy and Williard 2010; Stabenau et al. 1991; Stabenau and Vietti 2003; Stacy et al. 2016; Wilson et al. 2014). Although strict adherence to short tow times, availability of veterinary consultation, and other protective measures used during relocation trawling are intended to reduce the effects of capture, the additional concerns related to gravidity are not necessarily addressed by these measures. Therefore, we believe applying assumptions of the policy directive to gravid females subject to summer relocation trawling be a logical, risk-averse approach given the concerns and uncertainty related to the effects of capture on gravid females. Specifically, we will apply the directive's 10% post-interaction mortality rate for all female sea turtle captures in summer months in areas where they nest. Loggerhead sea turtles are the most likely gravid female sea turtles that will be encountered based on both the higher numbers of this species that nest in the action area and the higher numbers of this species that are reported captured in the action area. However, green and leatherback gravid females may also be captured. Kemp's ridley sea turtle nesting in the action area is very rare and will not be considered.

There is a general absence of data; however, on the number of female sea turtles captured by relocation trawlers in the action area due to the lack of relocation trawling during the summer months when nesting females may be present. Furthermore, relocation trawlers may also capture subadult and adult male sea turtles during any given project. Numerous studies have examined sex ratios of loggerhead sea turtle hatchlings, subadults, and adults, which demonstrate variance between year/study and region (Turtle Expert Working Group 2009); however, information on adult sex ratios is generally lacking for all regions in the Atlantic. The percentage of subadult female loggerheads ranges between 65-80% in various locations or assumed subpopulations, though this percentage apparently decreases between the immature and mature stages (Turtle Expert Working Group 2009). This information indicates that a large number of captures may not be mature turtles that could be gravid females. For comparison, we reviewed information provided by a private consulting firm (Coastwise Consulting) that performs relocation trawling both the action area for this Opinion and in the Gulf of Mexico. The reports provided from 2011-2018 included the size of turtles captured, which was compared to the size each species is expected to reach sexual maturity. Due to the lack of data on the estimated percent of adult females that may be in the action area, the review of relocation data considered all available projects and was not limited to only those occurring during summer nesting months. We believe that the information provided for the relocation trawling captures within the action area are the most appropriate to use to estimate future captures under the 2020 SARBO, except for green sea turtles. While no adult green sea turtles were captured in the projects reviewed, green sea turtles do nest within the action area and female green sea turtles may be encountered. Therefore, we will use the percent of adult green sea turtles captured in the Gulf of Mexico as a proxy. As discussed in Section 4.1.1.2 of this Opinion, green sea turtles in both the action area and the Gulf of Mexico belong to the same DPSs. Green sea turtle nesting occurs on sandy beaches throughout the southeastern United States between Texas and North Carolina, as well as Puerto Rico (Dow et al. 2007; NMFS and USFWS 1991). The percent of adults that may be captured are highlighted in Table 40. For the purposes of this Opinion, we assume that half of all adults captured are female since we do not have adequate information to determine the breakdown of the sex of turtles captured.

Table 40. Reported sea turtle captures between 2011-2018 – in the Atlantic Ocean (ATLO) and in the Gulf of Mexico (GOM)

Sea Turtles	ATLO Subadult Captures	ATLO Adult Captures	ATLO Total Captures By Species	ATLO Percent Adult	GOM Subadult Captures	GOM Adult Captures	GOM Total Captures By Species	GOM Percent Adult
Green	4	0	4	0.00%	41	8	49	16.33%
Kemp's ridley	39	1	40	2.50%	286	253	539	46.94%
Leatherback	0	12	12	100.00%	0	20	20	100.00%
Loggerhead	31	95	126	75.40%	68	551	619	89.01%
Total	74	108	182	59.34%	395	832	1227	67.81%

6.1.4.1.3 Estimated Relocation Trawling Take of Sea Turtles Under the 2020 SARBO

Similar to the analysis on the number of captures estimated to occur annually by hopper dredging discussed in Section 6.1 of this Opinion, we reviewed the total captures by species that occurred in the action area in the last 5 years and the last 21 years since the issuance of the 1997 SARBO. Since relocation trawling under the 1997 SARBO was limited, we also reviewed relocation trawling that occurred within the action area covered under separate Section 7 consultations, including areas dredged that will be maintained under the 2020 SARBO such as Savannah Harbor. Relocation has not occurred in the U.S. Caribbean and is not covered under this Opinion. Table 41 provides the information used to calculate the CPUE for relocation trawling based on the reported captures divided by the number of trawling tows completed during the same time period. The relocation trawling CPUE is then multiplied by the number of tows expected under this Opinion, as discussed in at the beginning of Section 6.1.4 of this Opinion, to estimate the number of reported captures that may occur annually under this Opinion. We determined that if the CPUE should be calculated on the full 21-year data set, due to the limited number of projects that had relocation trawling in the last 21 years, to most accurately estimate future relocation trawling take that may occur under the 2020 SARBO. Only 12 projects used relocation trawling within the action area in the past 5 years, in limited areas of the action area. We accordingly believe that relying on this limited number of projects, in limited parts of the action area, would not as accurately predict CPUE for relocation trawling as a result of the proposed action in the 2020 SARBO action area.

Since relocation trawling under the 1997 SARBO and other consultations completed in the action area was more limited than what will be implemented through the 2020 SARBO, we believe that the maximum amount of relocation trawling is a more accurate estimate of relocation trawling under the 2020 SARBO proposed action, and therefore of associated sea turtle take. Relying on the maximum trawling number is intended to account for the expected increase in relocation trawling in marine environments and the potential to catch higher numbers of species working in areas and at times not previously trawled, as well as increasing sea turtle populations. While 5,001 tows for a single project shown in Table 41 is higher than the rest of the trawling effort shown in Table 41, it is comparable to relocation trawling occurring in other areas outside of the action area. For example, in Mississippi, a single project conducted 563 days of relocation trawling using 2 relocation trawling vessels for a combined total of 14,762 tows (MisCIP). While this project was not a maintenance project, it does demonstrate the tow effort may increase if a project has a high density of ESA-listed species in the area and requires more than one relocation trawling vessel to operate at the same time. It should also be noted that of the relocation trawling that occurred in the action area (Table 13), both of the projects with the highest number of tows per project occurred in the last 2 years, and the number of relocation trawling tows per project appear to be generally increasing, indicating future projects are also likely to require additional tows to adequately relocate ESA-listed species in the area. The decision of where and when to dredge will continue to be adjusted using the risk-based assessment process, which may include adjustments to this list of dredging locations to maximize the amount of dredging that can be completed while minimizing take based on lessons learned for dredging and trawling at these locations. For example, the USACE plans to dredge some navigation channels outside of the North Atlantic right whale migration season (i.e., summer months), when sea turtles are expected to be present in greater densities. The navigation channels that the USACE intends to dredge in the summer months include Brunswick Harbor, Savannah Harbor, Charleston Harbor, Wilmington Harbor Entrance/Inner Ocean Bar, Morehead City, and the Manteo Entrance Channel. Dredging in the summer in these channels is expected to increase the number of tows needed and the estimated sea turtle captures that will be reported from relocation trawling. Because summer relocation trawling has been very limited historically, we do not have reliable information regarding the extent to which towing may be increased, but note that the number of tows at MisCIP included summer dredging that resulted in a high number of turtles being relocated during that time period. Thus, we believe that it is most appropriate to consider the trawling effort based on the maximum number of reported tows and maximum captures that may occur under this Opinion for estimating relocation trawling take of sea turtles, to account for the expected increase in relocation trawling tows associated with summer dredging, in areas where more sea turtles are expected to be present in higher densities, increased use of relocation trawling as a measure to minimize the impact of take, and increasing sea turtle populations. Therefore, we estimate the number of captures by relocation trawling based on the maximum towing effort expected under the 2020 SARBO and the species captured since the issuance of the 1997 SARBO shown highlighted in Table 41.

Table 41. Sea Turtle Relocation Trawling Calculations

	Total Tows	Green	Kemp's Ridley	Leatherback	Loggerhead
All Reports in Action Area (fiscal years 1997-2019)	31,595	53	91	25	358
Maximum per Project	5,001	29	19	10	90
CPUE (reported take/ total tows)		0.00168	0.00288	0.00079	0.01133
Maximum annual estimated take (CPUE x 155,031 tows)		260.06	446.52	122.67	1,756.64

While we expect the vast majority of relocation trawling captures to be nonlethal take, past projects have observed mortality associated with this activity. Therefore, we will assume a 0.38% mortality rate for captured sea turtles, as discussed in Section 6.1.4.1.1 of this Opinion, for all sea turtle captures in relocating trawling. Also, as discussed in Section 6.1.4.1.2 of this Opinion, we are including 10% post-interaction mortality for gravid females and the loss of a clutch by adult female turtles captured during the summer months associated with nesting season. The USACE stated that 6 channels (Brunswick Harbor, Savannah Harbor, Charleston Harbor, Wilmington Harbor Entrance/Inner Ocean Bar, Morehead City, and the Manteo Entrance Channel) may be dredged in the summer by hopper dredging that may also use relocation trawling to minimize the risk of lethal take, though other areas may also be included if they meet the PDCs of this Opinion. Based on limited availability of hopper dredges and the short timeframe (warmer months), no more than half of these 6 would be dredged in the summer in a single year. Therefore, we expect summer dredging to require relocation trawling equivalent to 15,003 tows (3 projects x 5001 maximum effort tows). Using all of this information, Table 42 calculates the estimated total annual lethal and nonlethal take estimated to occur for projects covered under this Opinion.

As discussed in this section, Kemp's ridley and hawksbill sea turtle nesting in the action area is rare and therefore we do not expect gravid females to be relocated that could result in post-interaction mortality or the loss of an egg clutch.

As discussed when estimating take from hopper dredging, the final estimated take covered under this Opinion for relocation trawling will be rounded to a whole number. However, rounding of take estimates will occur later in this analysis once all estimated take is calculated and combined in Section 6.1.5 of this Opinion. The final estimated take will also consider 3-year take average estimate discussed in Section 6 of this Opinion to account to variability, which will be calculated in a combined take estimate in Section 6.1.5. The annual take estimate will still be used to track take overall, but exceedance of take that would result in a need to reinitiate consultation will be based on the 3-year take average.

Table 42. Summary of Annual Estimated Sea Turtle Take from Relocation Trawling[74]

Row	Calculation	Green	Kemp's Ridley	Leatherback	Loggerhead
1	Annual estimated captures by species (Table 41)	260.06	446.52	122.67	1,756.64
2	Estimated CPUE (Table 36)	0.00168	0.00288	0.00079	0.01133
3	Estimated annual summer captures (Row 2 x 15,003 summer tows)	25.21	43.21	11.85	169.98
4	Estimated annual summer adult sea turtle captures (Row 3 x percent adult in Table 40)	4.12	1.08	11.85	128.17
5	Estimated annual adult female sea turtles captured in summer (Row 4 x 50%)	2.06	0.54	5.93	64.08
6	Annual post-interaction mortality of adult female sea turtles that may be gravid (Row 5 x 10% post-interaction mortality)	0.21	0.05	0.59	6.41

[74] As discussed in this section, Kemp's ridley and hawksbill sea turtle nesting in the action area is rare and therefore we do not expect gravid females to be relocated that could result in post-interaction mortality or the loss of an egg clutch.

Row	Calculation	Green	Kemp's Ridley	Leatherback	Loggerhead
7	Estimated sea turtles captured that are not adult female sea turtles that may be gravid in summer (Row 1 - Row 5)	258.00	445.98	116.74	1692.56
8	Annual post-interaction mortality of other sea turtles that are not gravid adult females(Row 7 x 0.38% post-interaction mortality)	0.98	1.69	0.44	6.43
9	Total annual estimated lethal take (Row 6 + Row 8)	1.19	1.75	1.04	12.84
10	Total annual estimated nonlethal take (Row 1 – Row 9)	258.87	444.77	121.63	1743.80
10	Total annual lost clutch (total female turtles captured in summer assumed to each lose 1 clutch- rounded up from row 5)	3	1	6	65

6.1.4.2 Effects to Sturgeon from Relocation Trawling

Atlantic Sturgeon

As previously discussed, 5 DPSs of Atlantic sturgeon were listed in 2012. While the reports in include Atlantic sturgeon captures prior to listing, these reports may not account for all captures as reporting the capture of this species was not required. In recent years, the number of reported Atlantic sturgeon captures in relocation trawling has increased, especially during winter months at the mouth of rivers where sturgeon are present (identified as sturgeon rivers in Appendix E). Because the 5 DPSs of Atlantic sturgeon were not listed until 2012, and reports of Atlantic sturgeon have been increasing over that time, we believe that the using the relocation trawling records since 2012 is the best available information for CPUE for this species. Though the risk-based assessment process will be used to try to minimize the number of take of ESA-listed species, relocation trawling will be used as a reasonable minimization measure to continue projects in these areas when sturgeon presence may be high, depending on the project and the anticipated risk to other ESA-listed species. While relocation trawling is done in conjunction with hopper dredging, trawling is not always necessary or appropriate depending on the location of the activity. Hopper dredging outside of sturgeon rivers is common; however, as work continues up into sturgeon rivers, the risk of take of sea turtles decreases, which may reduce the use of relocation trawling unless specifically needed to relocate sturgeon. Relocation trawling can also be harder to implement in these areas due to debris snags and the limited space to maneuver multiple pieces of equipment for projects in rivers. Reported sturgeon captures in were generally at projects located near the mouth of sturgeon rivers and the associated estuaries, rather than occurring further up in the rivers themselves where sturgeon are likely to be present in greater number. For these reasons, we believe that relocation trawling in sturgeon rivers is not likely to increase significantly, and it is more appropriate to consider the average number of relocation trawling tows as opposed to the maximum number of relocation trawling tows, for the purposes of calculating take. Therefore, we believe that 314.90 captures of Atlantic sturgeon may occur each fiscal year (Table 45). Since the DPS of sturgeon captured can only be identified by genetic testing, Atlantic sturgeon captured under the 2020 SARBO will be tested and the samples will be run to determine the DPS, as described in the PSO PDCs in Appendix H. Expected relocation trawling take by DPS is provided in Section 8.5 of this Opinion.

Of all reported Atlantic sturgeon captures by relocation trawling, only 1 lethal take was reported. This report lethal take was captured occurred in 2017 in Savannah when the net was retrieved with approximately 300 lbs of cannonball jellyfish and an unresponsive sturgeon. Despite attempts to resuscitate the fish by flushing water over its gills, it died. This rare instance demonstrates that mortality may happen, but that most captures are expected to be nonlethal. This 1 lethal report equates to a 0.34% mortality rate, which will be used to estimate sturgeon captures that may be lethal under this Opinion. Therefore, we estimate that 313.83 Atlantic sturgeon will be nonlethal captured in relocation trawling each year and 1.07 sturgeon captures will be lethal (314.90 estimated total take-1.07 estimated lethal take= 313.83 nonlethal captures).

Shortnose Sturgeon
We have no reports of shortnose sturgeon captured in relocation trawling. This is likely due to the limited amount of relocation trawling that has occurred in the area and that most relocation trawling occurs in the ocean, bays, and harbors where Atlantic sturgeon are more prevalent than shortnose sturgeon. As discussed in Section 4.1.3 of this Opinion, shortnose sturgeon are more properly characterized as "freshwater amphidromous," meaning that they move between fresh and salt water during some part of their life cycle, but rarely leave the rivers where they were born ("natal rivers"). However, in a recent report by the South Carolina Division of National Resources and Georgia Division of National Resources, the species was detected as far as 12.4 miles from the mouths of their spawning rivers in those states (Arendt et al. 2017). Based on their life history, shortnose sturgeon generally stay within rivers and typically are found further up river than hopper dredging under this Opinion will occur. In addition, the Sturgeon PDCs in Appendix E limit dredging in known aggregation areas to further protect sturgeon in these areas. Since we do not have relocation trawling records in rivers in the action area, but expect that they will be present in at least some locations where relocation trawling will occur, we rely on the estimated annual hopper take of shortnose sturgeon in Section 6.1.1 of this Opinion (estimated 1.69 shortnose sturgeon take annually), and assume that the same shortnose sturgeon that could be taken by hopper dredging in rivers or estuaries when they venture from rivers could also be relocated to minimize the risk of lethal take. Due to the low number of estimated captures and the low number of mortality observed by trawling of sturgeon (discussed above), we believe that all captures of shortnose sturgeon will be nonlethal.

6.1.4.3 Effects to Giant Manta Ray from Relocation Trawling

Giant manta rays are likely to be captured in relocation trawling in the action area, though we lack records of captures of this species to accurately estimate the number that may be captured. The lack of data is a result of the recent listing of this species under the ESA in 2018 (83 FR 2916, Publication Date January 22, 2018), that trawling in the action area has been limited (as previously discussed), and that prior reports of captures of rays were not accurately identified to know if they were giant manta rays. The best documentation that we have at the time of completion of this Opinion is from the northeast Atlantic, which is outside of the action area and in an area expected to have a lower abundance of this species than found in the action area. The reports from the northeast Atlantic are reports of mantas caught as bycatch in fisheries where NMFS' observers document each interaction with a Mobulid ray by species when possible. Observations historically included giant manta ray, Atlantic devil ray, unidentified ray, unidentified manta, and *Mobulidae* (any manta and devil ray species that could not be confirmed to species). Because of the unique form and cephalic lobes adjacent to the mouth of manta and devil rays it is unlikely that these records would have been listed more generally as an unidentified stingray or an unidentified ray; however, we do consider misidentification in these reports possible. Historically, many *Mobulidae* species may have been identified as giant manta rays because observers were provided with the Peterson's guide *Atlantic Coast Fishes* as a main source for identification, and the giant manta ray was the only large *Mobulidae* species shown. In 2015, NMFS Northeast Fisheries Science Center re-evaluated photo records of *Mobulidae* species and found that numerous historic records that were originally identified as giant manta rays were actually other *Mobulidae* species. Thus, historic records that did not include photos, or where photos were not detailed enough to determine a species, were then classified as an unidentified manta ray.

Based on the available unpublished data (NMFS Northeast Fisheries Observer Program data) from 2001-2015 of giant manta rays and unknown ray species captured in gear types used in the Northeast fisheries, we were able to estimate a CPUE based on the number of reported ray captures and the tow effort. The rays counted included those that were identified as giant manta rays through photo identification and others reported *Mobulidae* (any manta and devil ray species that could not be confirmed to species), assuming that they may have been giant manta ray. Table 43 shows take that may occur under this Opinion using the calculated CPUE and multiplying it by the estimated number of tows under this Opinion. We used the maximum number of tows estimated to occur annually under this Opinion (i.e., 155,031 tows) to account for the likelihood of encountering more giant manta ray in the action area than the reported captures in a fishery in the northeast. Giant mantra rays are year round residents in the action area for this Opinion, though a portion migrate to the Northeast during warmer months, including some that migrate out of the action area. In addition, new information suggests that giant mantas may occur in high numbers in specific areas within the action area where relocation trawling has historically not occurred. As discussed for other species take estimates, NMFS will continue to work with the SARBO Team to provide updated information on the presence of this species as part of the risk-based assessment process to further minimize take associated with this Opinion.

Table 43. Giant Manta Ray Estimated Relocation Trawling Captures

	2001-2015
Total tows	57,829.12
Total Captures	11
CPUE	0.000190
Maximum annual estimated take (CPUE x 155,031 tows)	29.49

We expect that all captures of giant mantra rays in relocation trawling will be nonlethal since they are large species that will be released directly from the trawling net according to the PSO PDCs in Appendix H. Giant mantra rays are large animals that are difficult to carry and maneuver and releasing them directly from the net will reduce the risk of harm to this species when captured. Due to the size and maneuverability of this species, we do not expect that they will be taken by hopper dredging and therefore releasing them back into the dredging area from relocation trawling is the safest option for this species.

6.1.4.4 Effects to Smalltooth Sawfish from Relocation Trawling

Smalltooth sawfish are known to be captured in nets and the long, toothed rostrum of a sawfish is prone to entanglement when captured in relocation trawling and it is difficult to disentangle it without harming the animal. Entangled animals frequently have to be cut free, causing extensive damage to nets. The entangled smalltooth sawfish can also endanger the crew if brought onboard a vessel. For these reasons, many historical records of smalltooth sawfish captures in fisheries note that they were either killed or released after their saws had been removed (e.g., (Bigelow and Schroeder 1953a; Evermann and Bean 1897; Henshall and United States Fish Commission 1891).

Due to the limited amount of relocation trawling that has previously occurred in the action area, particularly in Florida where smalltooth sawfish are more prevalent, we do not have any reported captures of this species in the action area in relocation trawling. However, we do have 5 reported captures in Tampa Bay (outside the action area) and other reports of captures in nets such as research studies and fisheries shown in Table 44. The reported captures north of Key West are also outside of the action area, but close enough to be relevant since the east side of the Florida Keys is within the action area. Of the 14 reported captures via relocation trawling, 6 were reported to be injured in the process, 3 of which reported broken rostrums (excluding the intentional illegal removal of a rostrum), and 8 reported being released alive with no other injuries reported. These reports demonstrate the risk of the rostrum being damaged in nets since 4 of the reported 14 captures had such an injury, which equates to 28.57% of the captures, which we use as an estimate to determine the number of individuals that may result in a lethal take since the fate of these animals after they are released is unknown.

The 4 smalltooth sawfish captured during relocation trawling in Tampa Bay served as an opportunity for NMFS to work closely with the PSO that handled the captures to help develop the appropriate PSO handling guidance for this species based on the information that they provided about the captures. We believe that smalltooth sawfish captured in relocation trawling under this Opinion will be released alive based on the requirements for them to be handled by a trained PSO and to cut the nets to release them. However, we also believe that some may have their rostrum damaged due to the fragility of this part of the body and the likelihood of it becoming entangled in the net.

We do not know the effort of trawling that resulted in the captures reported in Table 44, except for the unique case of the 5 relocation trawling captures in Tampa Bay. However, we believe that this could happen within the action area if trawling were to occur in an area where sawfish were aggregating as was the case with the 5 reported captures in relocation trawling all happening in Tampa Bay. The range of smalltooth sawfish is generally limited to Florida and relocation trawling will be limited in southeast Florida where sawfish are expected to be most prevalent (restrictions in southeast Florida are designed to be protective of ESA-listed corals). However, if sawfish are encountered, numerous captures could occur in the same area similar to the 4 captures during relocation trawling in Tampa Bay in 2019. In the 18 years of reported relocation trawling in the Gulf of Mexico, 5 smalltooth sawfish have been captured. This equates to a capture every 0.28 years (5 captures/18 years) or 1 capture every 3-4 years. We believe that a smalltooth sawfish could also be captured during relocation trawling in the action area covered under this Opinion at the same rate since the 2020 SARBO is expected to result in a similar rate of relocation trawling on the east coast of Florida as is being done on the west coast of Florida that resulted in these captures. Therefore, we assume that 1 sawfish capture will occur every 3 years and that 1 capture every 9 years could result in injury of a rostrum, which will be considered lethal take for the purposes of this analysis (28.57% of captures may be injured, which is approximately a third of the captures or 1 capture out of every 3 years of a single capture- 1 out of every 9 years).

Table 44. Smalltooth Sawfish Historic Captures in Trawling and Similar Nets

Date	Location	Number Captured	Disposition	Notes
8/12/06	Relocation trawling in Tampa Bay Entrance Channel	1	Released alive	Entangled in net, brought aboard to release, and returned to water in 18 minutes alive and swam away.
4/13/15	Florida Fish and Wildlife Conservation Commission Baitfish Cruise, south of Sanibel Island, Florida	1	Released alive	30 minute tow, in depths from 4.3 to 4.9 fathoms.
6/8/15	Shrimp fishery, Georgia (off Cumberland Island)	1	Released alive	Released in poor condition (sank upside down)
9/2/15	Shrimp fishery, northeast of Dry Tortugas, Florida	1	Released alive	Caught in net with the tail sticking out of TED and rostrum caught in net. Cut free from net and hoisted overboard using a whip line wrapped around the tail and winches. Sawfish released alive and moving per observer, but final disposition unknown.
7/19/16	Georgia Division of National Resources Turtle research trawl, east of Canaveral, Florida	1	Released alive	Entangled in the trawl net. Brought aboard to cut free from the net, whereupon it became very lively.
7/17/17	Shrimp fishery, Coastal Georgia	1	Released alive	Caught in net. Released with full rostrum attached, but break in rostrum from entanglement and thrashing in net, which caused the rostrum to bend/fold over on itself, presumably causing damage
12/13/17	Shrimp fishery, north northeast of Key West, Florida	1	Unknown	Crew unsuccessful in removing from net beside boat, brought aboard to cut out of nets. It was still showing some signs of life, but not sure if it was still alive when released. Water was too murky to see after it went under.
12/13/17	Shrimp fishery, north northeast of Key West, Florida	1	Unknown	Crew was not sure if it survived, but they retained approximately 16 inches of the broken off rostrum. Another fishing boat also reported catching a sawfish.
7/18/18	Shrimp fishery, northeast of St. Augustine, Florida	1	Dead	Witnesses report the shrimp trawl operator using a reciprocating saw to remove the rostrum. This incident was reported to Office of Law Enforcement to investigate it.
6/12/19	Shrimp fishery, north of Key West, Florida	1	Released alive	

Date	Location	Number Captured	Disposition	Notes
5/17/19-6/30/19	Relocation trawling, Tampa Bay Entrance Channel Dredging	4	Released alive	Four relocation trawling captures (6/12/19, 6/24/19, and 2 on 6/26/19). All were entangled in net, but released by cutting the net and not bringing the animal aboard. One retrieved with a broken rostrum.

6.1.5 Total Estimated Captures of ESA-listed Species by Hopper Dredging, Cutterhead Dredging, and Relocation Trawling

As discussed earlier, relocation trawling is used as a measure to reduce the lethal take associated with hopper dredging, thereby minimizing the impact of take by the proposed action. However, this measure does also result in take of those animals captured. We have evaluated the effects of relocation trawling and determined that use of relocation trawling will result in nonlethal and, in limited instances, lethal take. This section combines the estimated nonlethal and lethal take associated with both hopper dredging and relocation trawling by species estimated to occur both annually and on a 3-year rolling average to account for variability in captures, discussed in Section 2.9.3.1 of this Opinion. Table 45 summarizes the take estimate by species discussed in Section 6.1 of this Opinion for hopper dredging and in Section 6.1.4 of this Opinion for relocation trawling, including the estimated observed and unobserved take estimated to occur annually and the proportion of take that is expected to be lethal and nonlethal. The estimated take is then rounded to the next whole number since a fraction of a species cannot be captured. Rounding up also provides an estimate that is conservative towards the species to evaluate the risk of these actions jeopardizing the species, which will be evaluated in Section 8 of this Opinion.

Table 46 combines both hopper dredging and relocation trawling take estimates from Table 45 by those that are will be observed, unobserved, lethal, and nonlethal take both annually and on a 3-year basis. As stated earlier in this section, a 3-year rolling average of observed take will be used to determine if the effects analyzed in this Opinion are accurate, and if reinitiation of consultation is necessary on the basis of exceeding the ITS, to account for the expected variability in take annual levels discussed earlier in this section.

We believe that even though relocation trawling used as a minimization measure to reduce lethal take associated with hopper dredging will in and of itself result in lethal take in limited instances, the use of relocation trawling minimizes the risk of harm from hopper dredging and is an appropriate minimization measure. The PDCs in this Opinion are designed to maximize the observability of take associated with hopper dredging and to minimize the risk of lethal take, from relocation trawling and hopper dredging, including providing PSO guidance on how to perform their role on both hopper dredges and relocation trawlers. We believe that adherence to the PSO PDCs will continue to reduce harm from relocation trawling by ensuring PSOs understand the requirements of how to properly handle species captures. The PSO PDCs also will be updated as needed to ensure that the handling guidance continues to include the best available methods to reduce harm to listed species.

Table 45. Total Estimated Take of All Mobile ESA-listed Species from Hopper Dredging, Cutterhead Dredging (sturgeon only), and Associated Relocation Trawling

Species	Annual Non-Lethal Observed Take	3-Year Non-Lethal Observed Take	3-Year Non-Lethal Observed Take (Rounded Up)	Annual Lethal Observed Take	3-Year Lethal Observed Take	3-Year Lethal Observed Take (Rounded Up)	Annual Lethal Unobserved Take	3-Year Lethal Unobserved Take	3-Year Lethal Unobserved Take (Rounded Up)
Relocation trawling									
Green sea turtle	260.06	780.18	781	0	0	0	1.19	3.57	4
Kemp's ridley sea turtle	446.52	1,339.56	1,340	0	0	0	1.75	5.25	6
Leatherback sea turtle	122.67	368.01	369	0	0	0	1.04	3.12	4
Loggerhead sea turtle	1,756.64	5,269.92	5,270	0	0	0	12.84	38.52	39
Atlantic sturgeon	313.83	941.49	942	1.07	3.21	4	0	0	0
Shortnose sturgeon	1.68	5.04	6	0	0	0	0	0	0
Giant manta ray	29.49	88.47	89	0	0	0	0	0	0
Smalltooth sawfish	0	1	1[75]	0	0	0	0	0	0
Hopper Dredging									0
Green sea turtle	0	0	0	19.73	59.18	60	19.73	59.18	60
Kemp's ridley sea turtle	0	0	0	18.60	55.80	56	18.60	55.80	56
Leatherback sea turtle	0	0	0	0	0	0	0	0	0
Loggerhead sea turtle	0	0	0	34.38	103.14	104	34.38	103.14	104
Unidentified sea turtle	0	0	0	2.25	6.76	7	2.25	6.76	7
Atlantic sturgeon	0	0	0	21.98	65.94	66	21.98	65.94	66
Shortnose sturgeon	0	0	0	1.69	5.07	6	1.69	5.07	6
Giant manta ray	0	0	0	0	0	0	0	0	0
Smalltooth sawfish	0	1	1	0	0	0	0	0	0
Cutterhead Dredging									
Atlantic sturgeon	0	0	0	0.08	0.24	1	0	0	0
Shortnose sturgeon	0	0	0	0.45	1.35	2	0	0	0

[75] 1 total per 9 year period (observed injuries during relocation trawling assumed to result in post-interaction mortality),

Table 46. 3-Year Total Estimated Nonlethal and Lethal Take from Hopper Dredging, Cutterhead Dredging, and Relocation Trawling.

Species	3-Year Non-Lethal Observed Take	3-Year Lethal Observed Take	3 Year Lethal Observed Unidentified Sea Turtles Total[76]	3-Year Lethal Observed Total (Identified + Unidentified [sea turtles])	3-Year Lethal Observed Total (Rounded Up)	3-Year Lethal unobserved Total (Identified + Unidentified [sea turtles])	3-Year Lethal Unobserved Take (Rounded Up)	3-Year Lethal Observed + unobserved	3-Year Sea Turtle Lost Egg Clutch
Green sea turtle	781	59.18	1.83	61.01	62	61.01	62	124	3
Kemp's ridley sea turtle	1340	55.80	1.73	57.52	58	57.52	58	116	1
Leatherback sea turtle	369	0	0	0	0	3.12	4	4	6
Loggerhead sea turtle	5270	103.14	3.19	106.33	107	106.33	107	214	65
Atlantic sturgeon	942	71	N/A	71	71	65.58	66	137	N/A
Shortnose sturgeon	6	8	N/A	8	8	6	6	14	N/A
Giant manta ray	89	0	N/A	N/A	0	0	0	0	N/A
Smalltooth sawfish	1	0	N/A	N/A	1[77]	N/A	0	N/A	N/A

[76] Section 6.1.2 includes a discussion outlining how the unidentified sea turtle (Table 36) allocation was determined.
[77] 1 total per 9 year period (includes observed injuries during relocation trawling assumed to result in post-interaction mortality.

6.2 Effects to Johnson's Seagrass

NMFS believes these proposed actions are likely to adversely affect Johnson's seagrass, which is listed as threatened under the ESA. However, no ITS or reasonable and prudent measures will be issued, because the ESA does not require Biological Opinions to contain ITS for plants, only for fish and wildlife species (USFWS and NMFS 1998).[78]

As discussed in Appendix D, we expect Johnson's seagrass to be removed during maintenance dredging activities covered under this Opinion. The USACE provided information about the estimated loss of Johnson's seagrass that may occur as a result of activities covered under this Opinion. Their review considered the maintenance dredging in the IWW (maintaining the 12 ft depth and 125 ft wide channel) and associated channels including a review of information available on past dredging events, seagrass surveys completed within the range of Johnson's seagrass, and anticipated dredging in this area that would be covered under this Opinion. It is important to note that we believe the majority of Johnson's seagrass removal will occur while maintenance dredging the IWW, which is excluded from Johnson's seagrass critical habitat. However, the loss of the species in this area must still to be evaluated.

In SARBA Appendix M, the USACE stated that, based on their analysis, dredging in these areas has shown that when seagrasses are present and are disturbed they re-colonize and persist even though cyclical impacts have occurred. Pre and post-construction surveys show seagrass within these areas quickly recolonizes following the dredging events. Specifically, the USACE reviewed pre- and post-survey results from 1999 to 2015 provided in an analysis for the USACE SAJ permit SAJ-93 that is used for navigation maintenance dredging of the IWW by the Florida Inland Navigational District. Based on this review, 15.35 acres of Johnson's seagrass were directly removed by dredging between 1999 and 2015 within the range of Johnson's seagrass, of which 5 acres of loss occurred in 2014 and 6 acres of loss occurred in 2015. In 2014, the surveys showed that the number of acres of seagrass loss equaled the number of acres where seagrass regrew in areas previously maintenance dredged. Therefore, the USACE estimates that an average of 5-6 acres of Johnson's seagrass will continue to be removed by maintenance dredging each year. Because the IWW is routinely dredged, we believe that the removed Johnson's seagrass there is likely to be a cycle of regrowth and dredging in this area.

NMFS completed an ESA Section 7 consultation (NMFS 2001a) that evaluated the effects of maintenance dredging of ports and the IWW within the range of Johnson's seagrass critical habitat under USACE permit SAJ-93 discussed above. This 2001 consultation encouraged, but did not require, pre- and post-construction surveys to document the presence of seagrasses within and adjacent to maintenance dredging areas and to document the loss of seagrasses associated with the project. The limited information collected and provided by the USACE is documented below. The specific seagrass surveys and their results were not provided due to the inconsistent documentation of that information provided to NMFS and the varying time periods before and after dredging events that limited the ability to directly compare the loss of seagrasses that occurred as a result of dredging. The report did highlight that seagrasses in these shallow water environments are dynamic and often regrow in dredged areas or areas adjacent to dredging. Therefore, we will use the USACE estimate of up to 6 acres of Johnson's seagrass will be

[78] *See, e.g., Ctr. for Biological Diversity v. Bureau of Land Mgmt.*, 833 F.3d 1136, 1145 (9th Cir. 2016)

removed by projects covered under this Opinion annually, which will be analyzed further in the Jeopardy Analysis in Section 8.9 of this Opinion. This area will all occur in the IWW with different sections being dredged each year.

Our review of literature used in the USACE SARBA analysis in Appendix M (USACE 2017) provides more detail that stated:

- Areas not maintenance dredged since 1999: Broward Reach I, St. Lucie Reach I, St Lucie Reach II, and Brevard Reach II.
- Areas routinely maintenance dredged every few years: Baker's Haulover, Palm Beach P-31, Palm Beach P-50, Jupiter, Crossroads, and Brevard/Haulover Canal.
- Maintenance dredging areas where seagrasses may occur: The USACE stated that seagrasses are not typically found in the maintenance dredging channel footprint, except for specific sites listed below. Therefore, Johnson's seagrass may be directly removed from these locations.
    - Areas where the channel is positioned close to shallow areas, or where the channel is shallow: Discontinuous seagrass is observed in a few isolated shallow areas, such as Indian River, St Lucie, Martin, and Broward Counties.
    - Areas near coastal inlets where water clarity is very good: The distribution and abundance of seagrass is increased in these areas. Inlets are the most dynamic coastal areas and sediment transport which causes shoaling and natural burial of seagrass can also result in potential colonization by pioneer seagrass species; these areas are generally on an active maintenance dredging cycle. Crossroads, Jupiter (Cuts P-1 to P-4), Palm Beach Cut P-50, and Baker's Haulover are proximate to coastal inlets where shoaling frequently occurs with subsequent colonization by pioneer seagrass species within or very close to the channel.
    - St. Lucie County: St. Lucie Reach 1 dredging project identified seagrasses within Cut SL-5. As of the publishing of the study, Cut SL-6 was being reviewed by the USACE for permit compliance as it is suspected that the City of Ft. Pierce's storm protection island system constructed between April 2012 and March 2014 contributed to massive shoaling and subsequent seagrass colonization in this portion of the IWW. This portion of the IWW has not been maintenance dredged since its original construction to the authorized depth in the 1960's and there is no record of the navigation channel in this area supporting seagrass or shoaling prior to the recent shoaling event following construction of the islands.
    - Martin County: The Martin County, Crossroads is the area at the confluence of the IWW and Okeechobee Waterway near St. Lucie Inlet, which has been dredged on an average 3-year basis since 1963. High current velocities associated with the inlet possibly destabilize sediments near the channel and account for the restricted distribution of seagrass within this highly dynamic area. Based on a survey conducted in August and September 2002, 2 patches of seagrass totaling 0.11 acre (*H. wrightii, H. johnsonii,* and *S. filiforme*) were located within 50 ft of the navigation channel. The post-survey report for this project stated there were no visible effects of dredging (e.g. anchor holes, pipeline placement, etc.) documented during the October 2013 survey. However, the January

2014 post-construction report concluded an 11% reduction of seagrass outside the dredge area compared to the pre-construction survey, assumed to be from the summer 2013 releases from Lake Okeechobee's nutrient rich fresh water via the St. Lucie Canal to the St. Lucie River decreased salinity and increased bluegreen algal growth and production.

- Palm Beach County, Port of Palm Beach IWW deepening project area: Surveys for this project identified 5.82 acres of seagrass within the proposed dredge area with this area dominated *H. decipiens* with some *H. johnsonii* and *H. wrightii*. The proximity to the Lake Worth Inlet and inflows from the Gulf Stream Current make this area unique as it contains very good water clarity and water quality.
- Broward County IWW Deepening Project Area: Johnson's seagrass was most abundant in shallow water along the margin of the IWW to depths of 8 ft. The limits of the survey did not include the actual navigation channel, however, in many cases seagrass beds were well landward of the edge of the channel.
- Broward County, Port Everglades: Multiple years of seagrass surveys in this area did not identify seagrasses in the channel. Species present adjacent to the channel included *H. wrightii, H. decipiens*, and *H. johnsonii*. The 2009 report showed an overall decrease in seagrass coverage in the northern area of the project and an expansion of seagrass in the southern area of the project.
- Palm Beach County: A high occurrence of seagrass exists at the northern end of the project area (Cut P-1) and drops sharply in quantity at the southern end (Cut P-5 to P-9). Numerous pre- and post-construction surveys were cited in the study documenting loss of seagrasses and areas where seagrasses had regrown, including Johnson's seagrass.
- Miami-Dade: Multiple years of seagrass surveys identified seagrasses in portions of the channel in the Baker's Haulover areas, though none of the surveys identified Johnson's seagrass. Also the Baker's Haulover borrow areas surveys did not identify seagrasses on this shoal.

6.3 Effects to ESA-Listed Corals

We expect take of ESA-listed corals that are relocated from areas within 100 ft of the ETOF for beach nourishment projects and within the footprint of pipeline corridors within the range of ESA-listed corals, as discussed in Section 3.2.2 of this Opinion. The available information on the beach nourishment and pipeline projects that have previously occurred in the action area in the last 12 years are summarized in Table 47and project locations that we reviewed are also shown in Figure 37 -Figure 48 and discussed in more detail in the analysis that follows. We expect that the projects identified in Table 47 (i.e. previous beach nourishment projects in the action area) are representative of projects that will occur under 2020 SARBO.

This Opinion only covers existing pipeline corridors and beach nourishment within the range of ESA-listed corals as follows:

- Projects are limited to those occurring within the existing fill template for beach nourishment projects and within the existing pipeline corridor and therefore any corals found in these areas are limited to those that recruited back into the area since the last nourishment or pipeline project (Coral PDC Section 2.3 and Section 2.4 in Appendix C).

- Beach nourishment and pipeline projects are limited to South Florida. These projects in the U.S. Caribbean are not covered under this Opinion (Coral PDC Section 2.3 and Section 2.4 in Appendix C).
- ESA-listed corals that may be found within beach nourishment or pipeline projects are limited by the range of the species provided in Table 21. For example, elkhorn corals are only in Southeast Florida from the northern limit of Broward County to the Dry Tortugas; Puerto Rico; U.S. Virgin Islands. Since projects in the U.S. Caribbean are not covered under this Opinion, we expect that the only elkhorn coral that may recruit back into the footprint of a previously filled beach nourishment or previously used pipeline project is limited to Broward and Miami-Dade County, Florida, and based on the location information provided, no projects are expected in the Florida Keys.
- However, we assume that no pillar or rough cactus coral colonies occur in pipeline corridors because of diseases in the area that continue to reduce already low densities of these corals in the action area, as reflected in Table 48.
- Since the corals that may be need to be relocated under this Opinion are limited to those that have recruited back into a previously constructed area, we believe that pillar or rough cactus coral colonies will not be found. These species already occur in low densities in the area where these projects will occur and have been further reduced by coral diseases in the area. Therefore, pillar and rough cactus coral will not be considered in this Section.

Table 47. ESA-Coral and Hardbottom Information from Previous Beach Nourishment Projects (2008-2017)[79]

Project Location	County	Project Date	Beach Nourishment Location Information	Pipeline Corridor Information
Jupiter Carlin	Palm Beach	2013	No corals identified in vicinity	No information available
Midtown and Phipps Ocean Park	Palm Beach	2013	No corals identified within 500 ft of ETOF. Hardbottom identified 20-100 ft from ETOF except adjacent at R-100 (NMFS 2014)	No information available
Ocean Ridge	Palm Beach	2013	No *Acropora corals* identified and no information on the presence of other corals in the vicinity. Hardbottom identified 3,000 ft from ETOF. (NMFS 2014)	Pipeline used but no information about presence of corals.
Delray Beach	Palm Beach	2013	No corals identified within 500 ft of ETOF. No hardbottom identified near ETOF.	no corals in the project area (5 Batched Palm Beach Shore Protection Projects- informal 2013)
Boca Raton	Palm Beach	2013	No information available	Pipeline corridor located between hardbottom areas (NMFS 2014)
Segment 2	Broward	2013	Corals identified within 492 ft from hardbottom edge= 1 elkhorn and 13,153 staghorn corals. We estimated 1,232 staghorn colonies within 100 ft of ETOF (see Figure 38 -Figure 45); 1 elkhorn colony and 15 staghorn colonies previously relocated from ETOF where it crossed hardbottom; no other ESA corals identified. (Coastal Planning & Engineering 2013), Figure 2a-h	Sand from upland source, so no pipeline was used
Segment 3	Broward	2011	287 staghorn corals within 302 ft from hardbottom edge (see Figure 47 - Figure 48); estimate none are within 500 ft of ETOF (NMFS 2011b, Figures 3-5)	No information available
Segment 3	Broward	2017	Staghorn corals observed (Figure 46 - Figure 48); no other ESA-listed corals observed. Hardbottom identified within 200-300 ft from ETOF (NMFS 2011b, Figures 3-5).	No information available
Sunny Isles	Miami-Dade	2011	No corals identified in vicinity. Hardbottom identified within 310 ft of ETOF.	No information available
Sunny Isles	Miami-Dade	2015	No corals or hardbottom within 750 ft of ETOF.	No information available
Bal Harbour	Miami-Dade	2011	Coral identified 830 ft from ETOF. Hardbottom identified within 310 ft of ETOF.	No corals identified in pipeline corridor
Bal Harbour	Miami-Dade	2015	No corals identified in vicinity. Hardbottom identified within 580 ft from shore.	No information available

[79] All information was provided by the USACE, unless otherwise noted. Projects are listed north (starting at the top) to south (at the bottom of the table).

Project Location	County	Project Date	Beach Nourishment Location Information	Pipeline Corridor Information
Surfside and Miami Beach	Miami-Dade	2008	No corals or hardbottom identified in vicinity	No information available
Miami Beach, Contract E	Miami-Dade	2012	No staghorn corals adjacent to nourishment site; no coral hardbottom near nourishment site (NMFS 2009b)	approximately 3,783 ft of pipeline over hardbottom; density staghorn corals 0.055 colonies per m^2 (NMFS 2009b); 68 staghorn corals relocated out of the pipeline corridor (DERM 2012)
Miami Beach	Miami-Dade	2015	No corals or hardbottom identified within 750 ft of ETOF	No information available

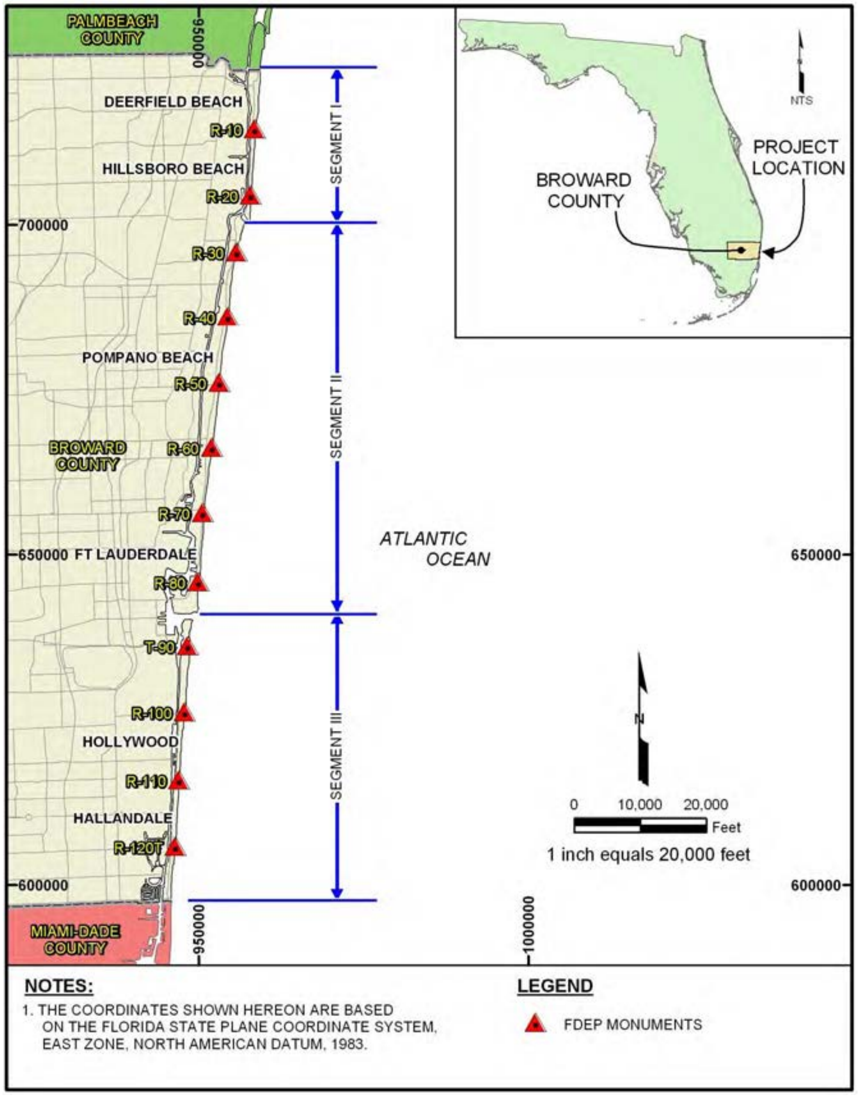

Figure 37. Broward County, Florida Beach Nourishment Segments 1, 2, and 3.
Image from (Coastal Planning & Engineering 2013).

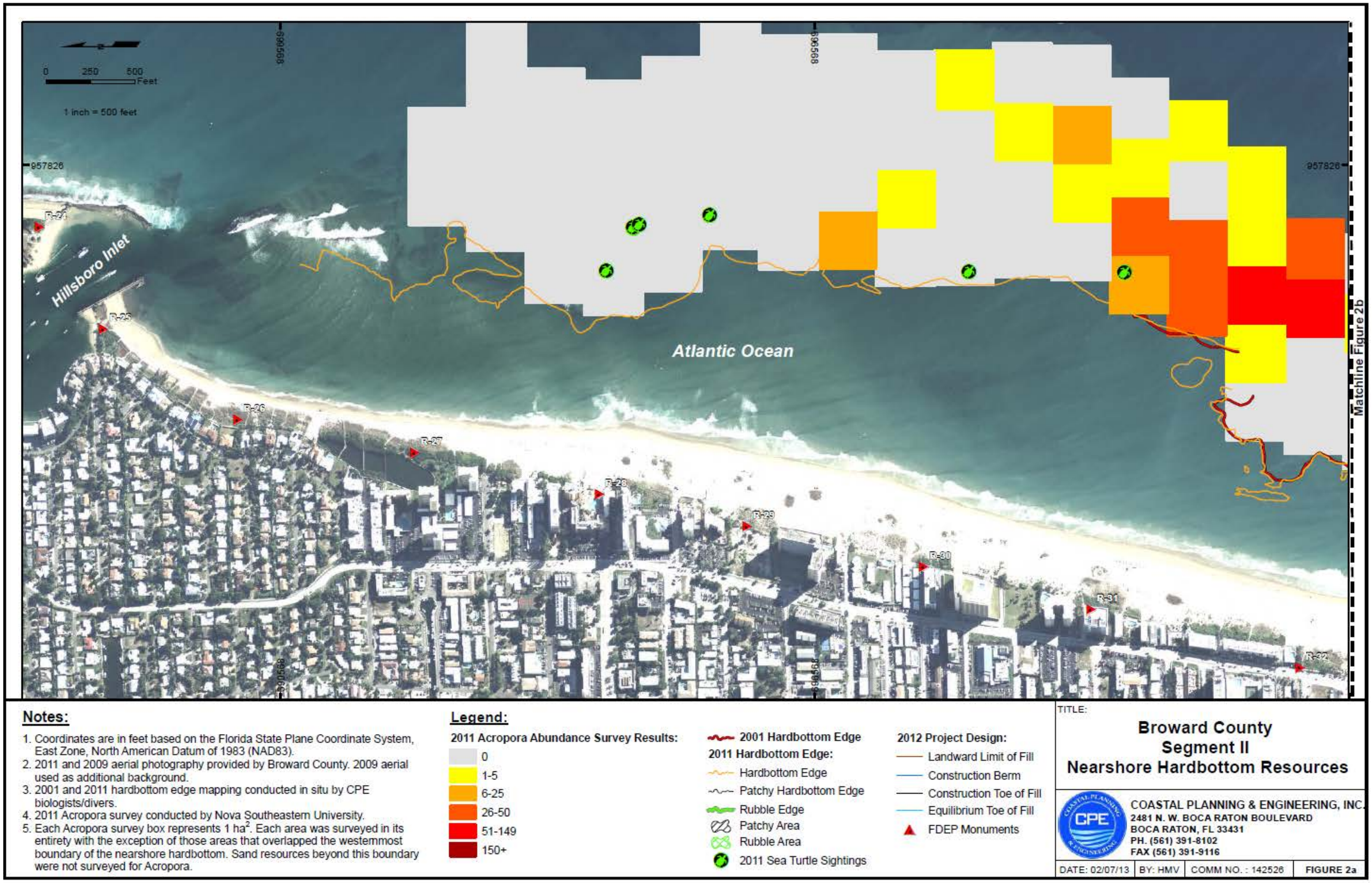

Figure 38. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys.
Image from (Coastal Planning & Engineering 2013).

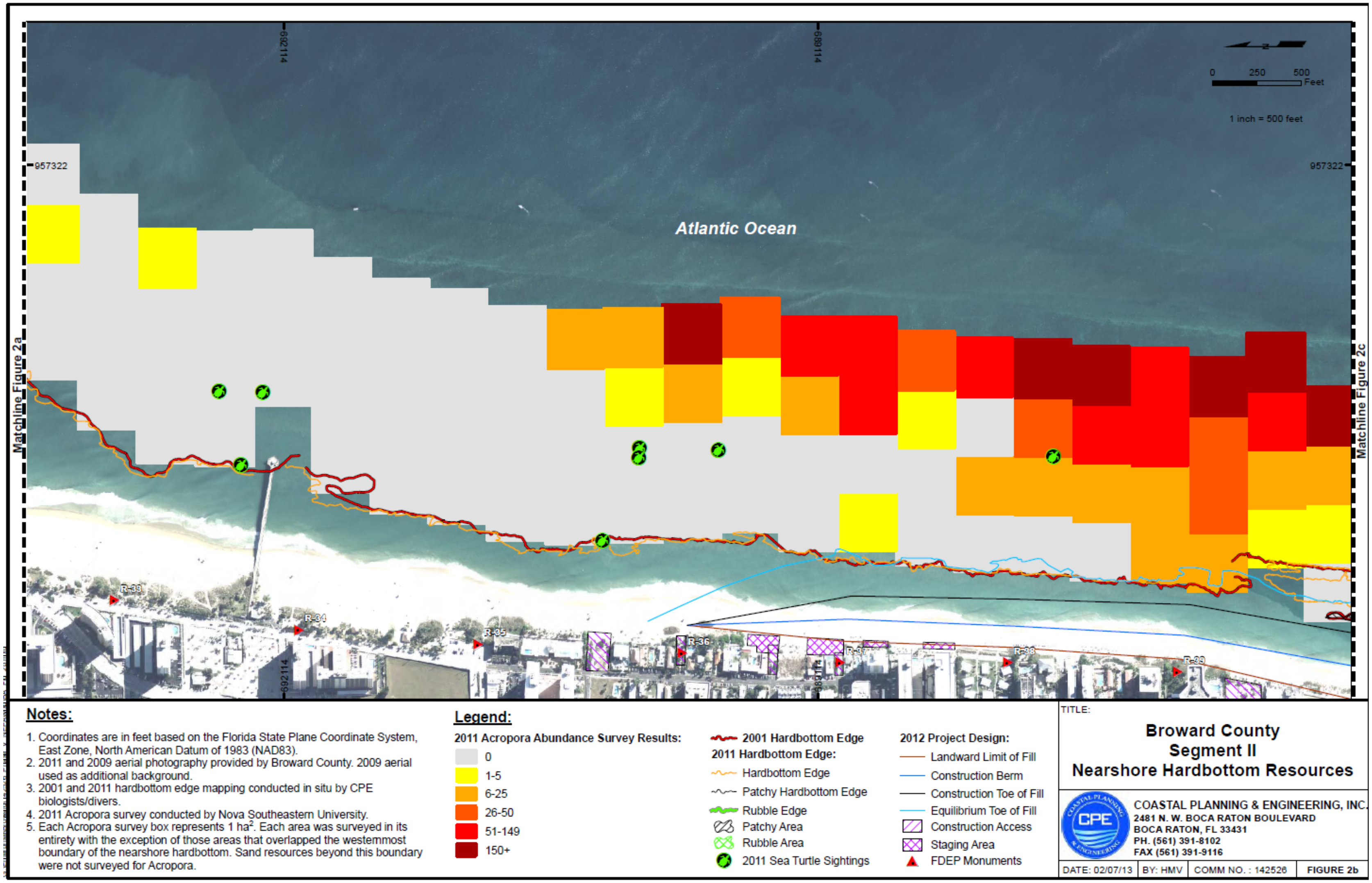

Figure 39. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys. Image from (Coastal Planning & Engineering 2013).

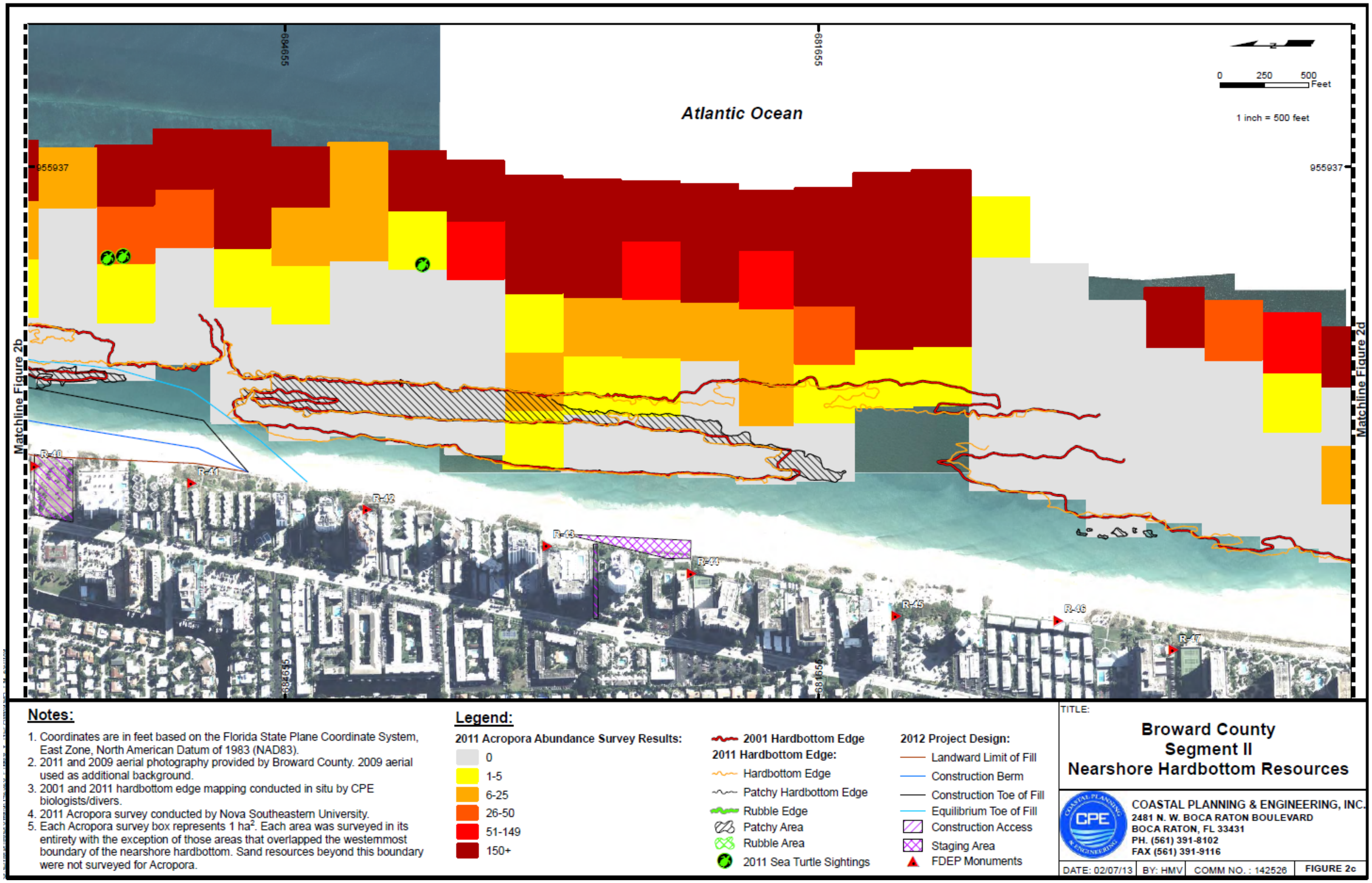

Figure 40. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys. Image from (Coastal Planning & Engineering 2013).

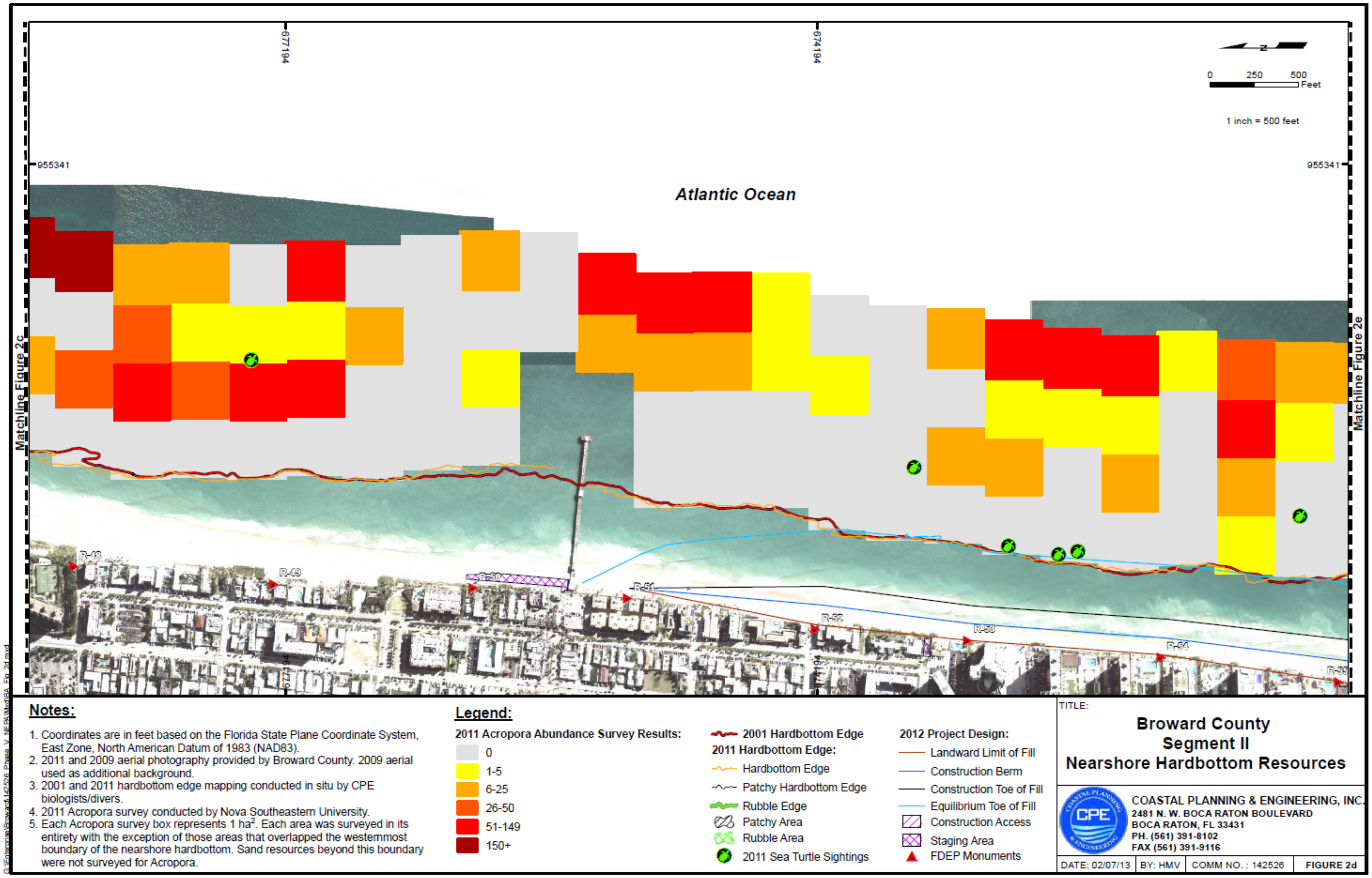

Figure 41. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys. Image from (Coastal Planning & Engineering 2013).

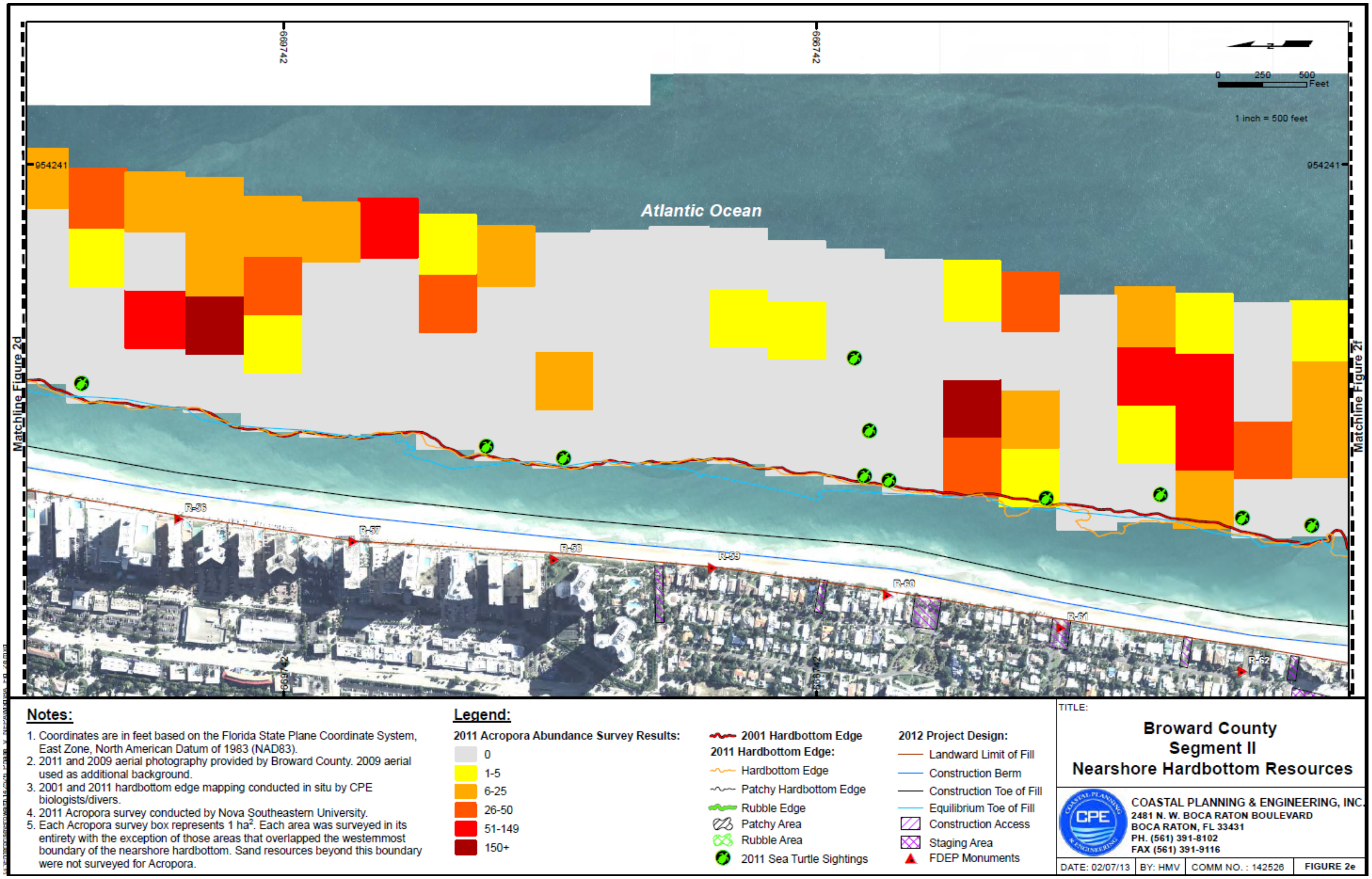

Figure 42. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys.
Image from (Coastal Planning & Engineering 2013).

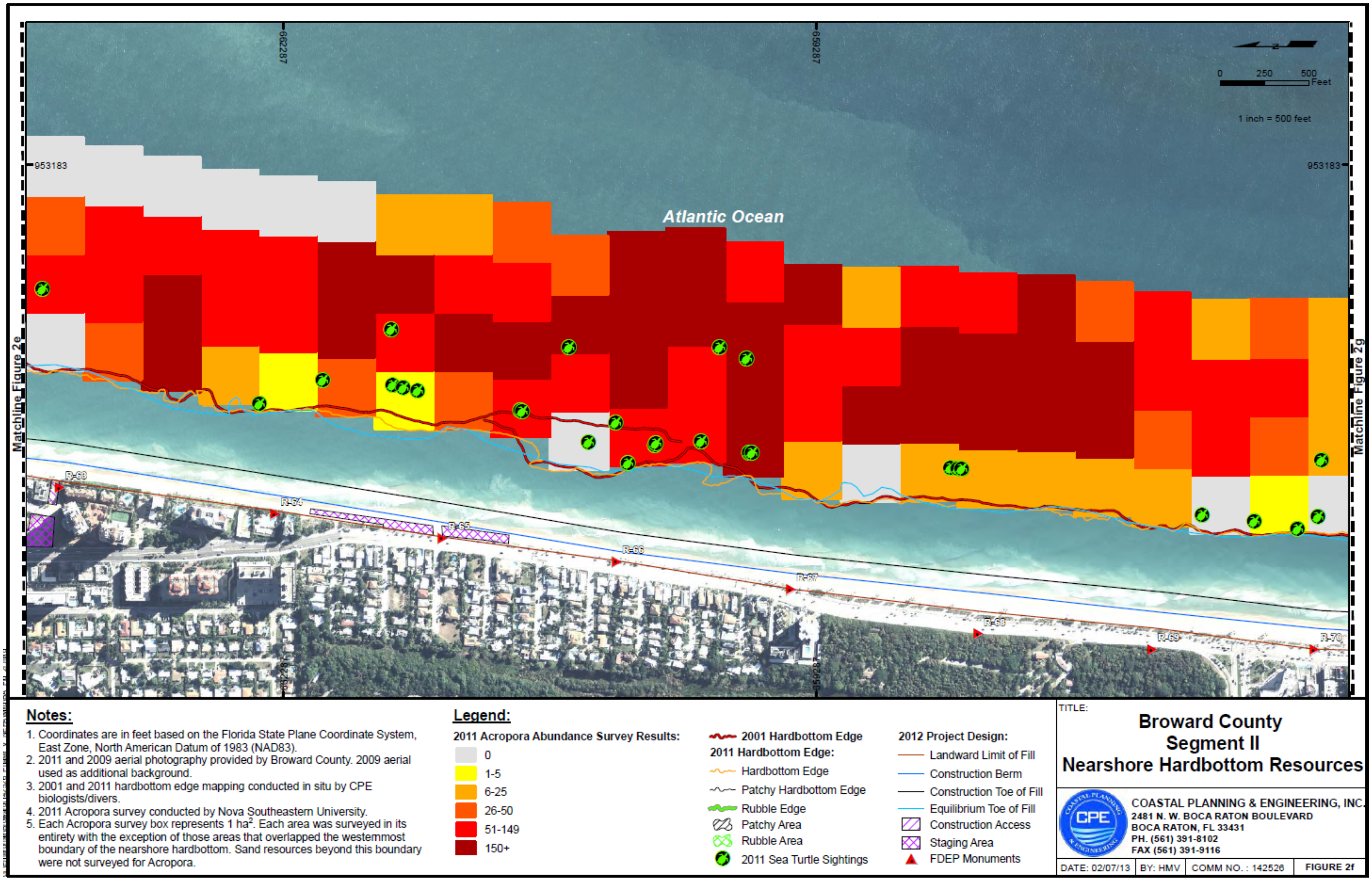

Figure 43. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys. Image from (Coastal Planning & Engineering 2013).

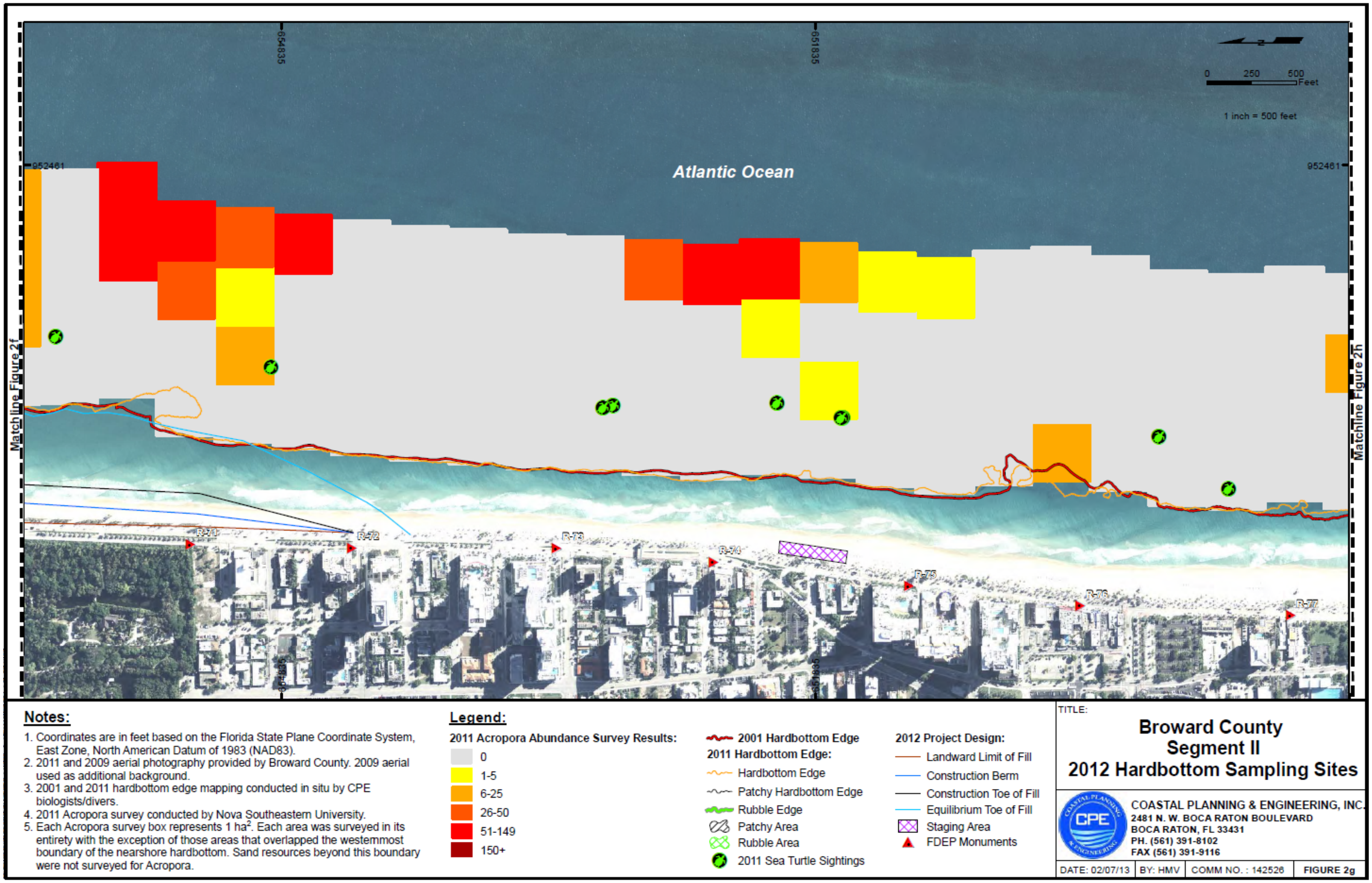

Figure 44. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* surveys.
Image from (Coastal Planning & Engineering 2013).

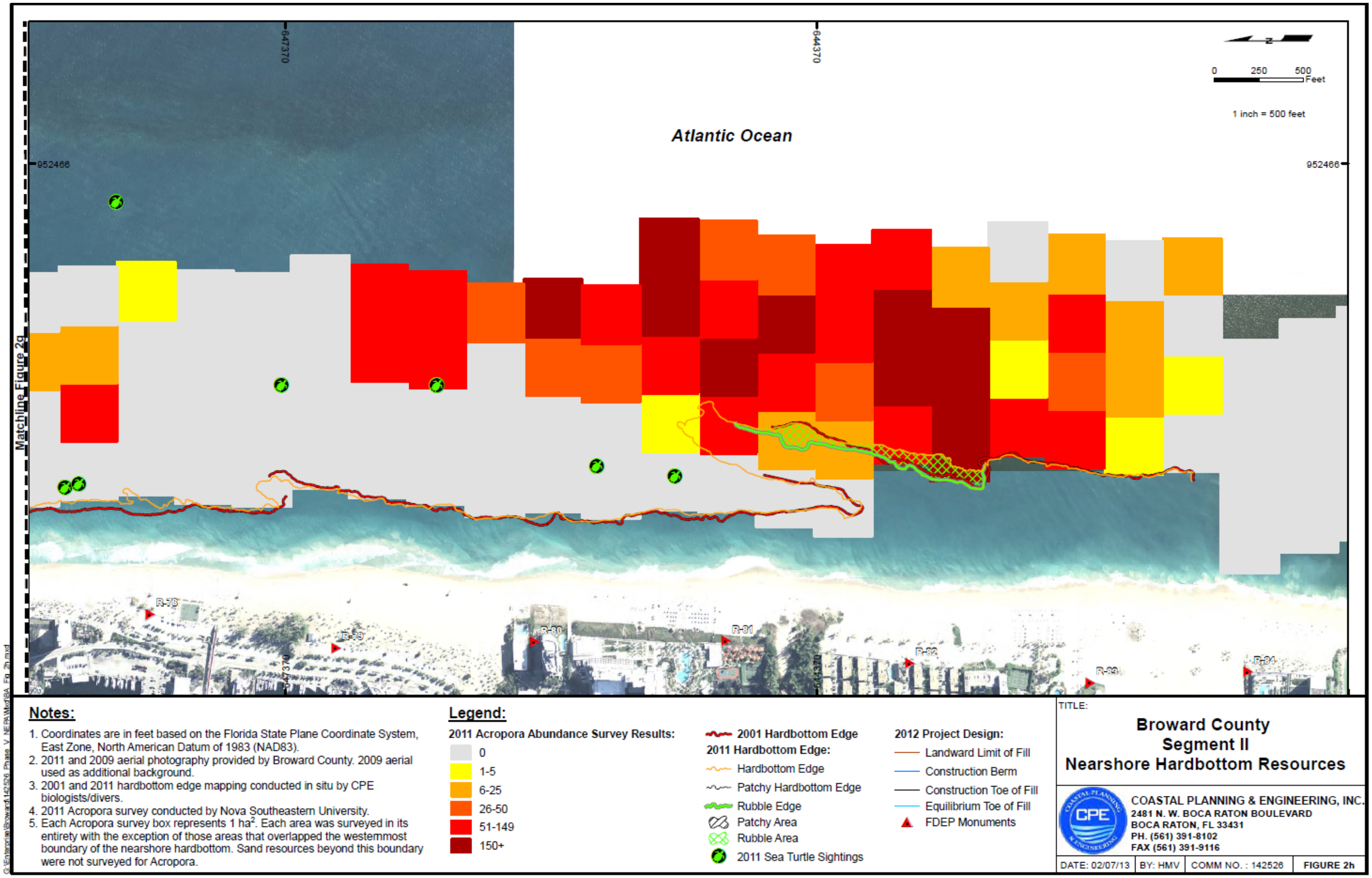

Figure 45. Broward County Segment 2. ETOF, hardbottom edge, and *Acropora* survey.
Image from (Coastal Planning & Engineering 2013).

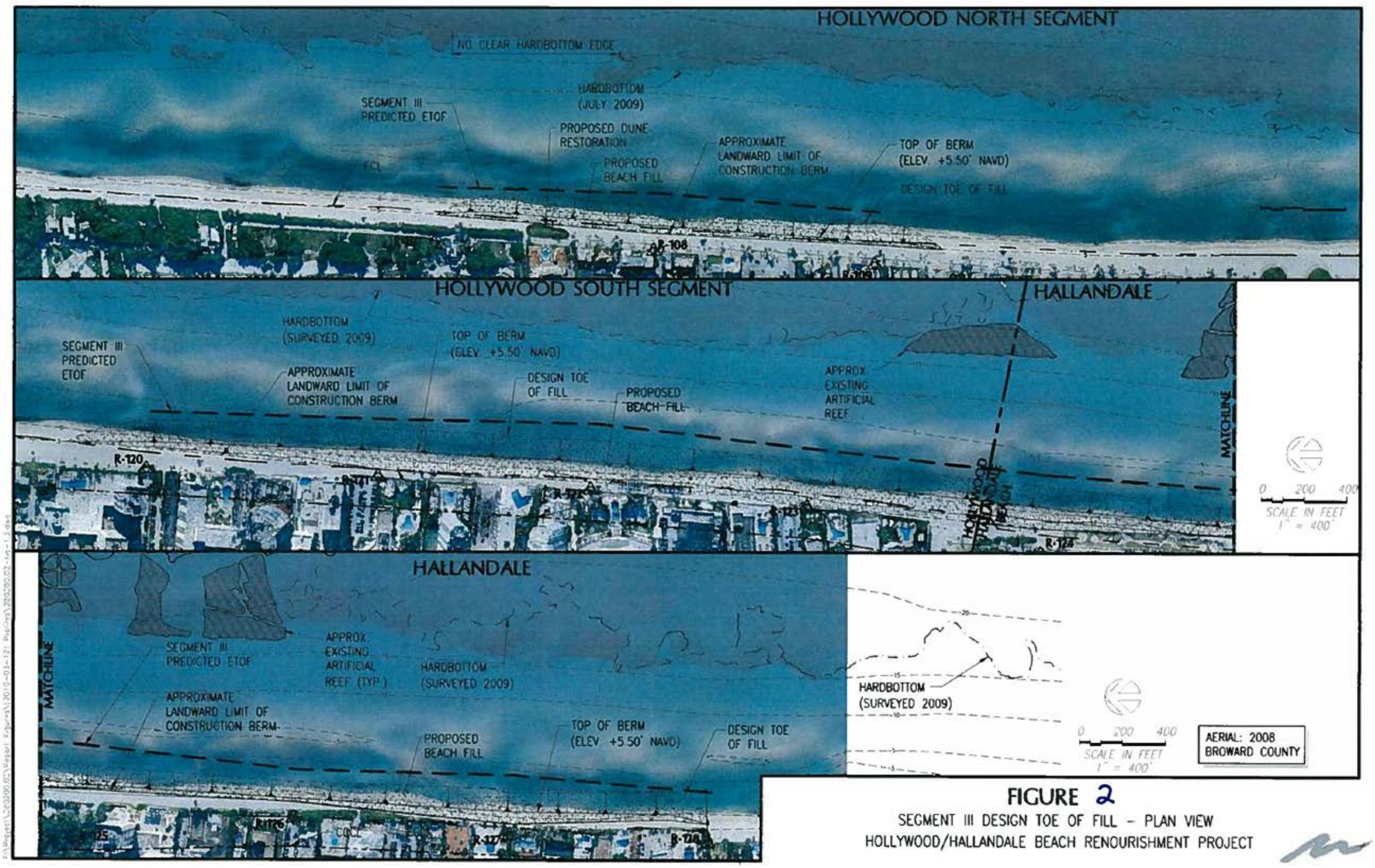

Figure 46. ETOF in Broward Segment 3, Hollywood/Hallandale Beach (NMFS 2011b).

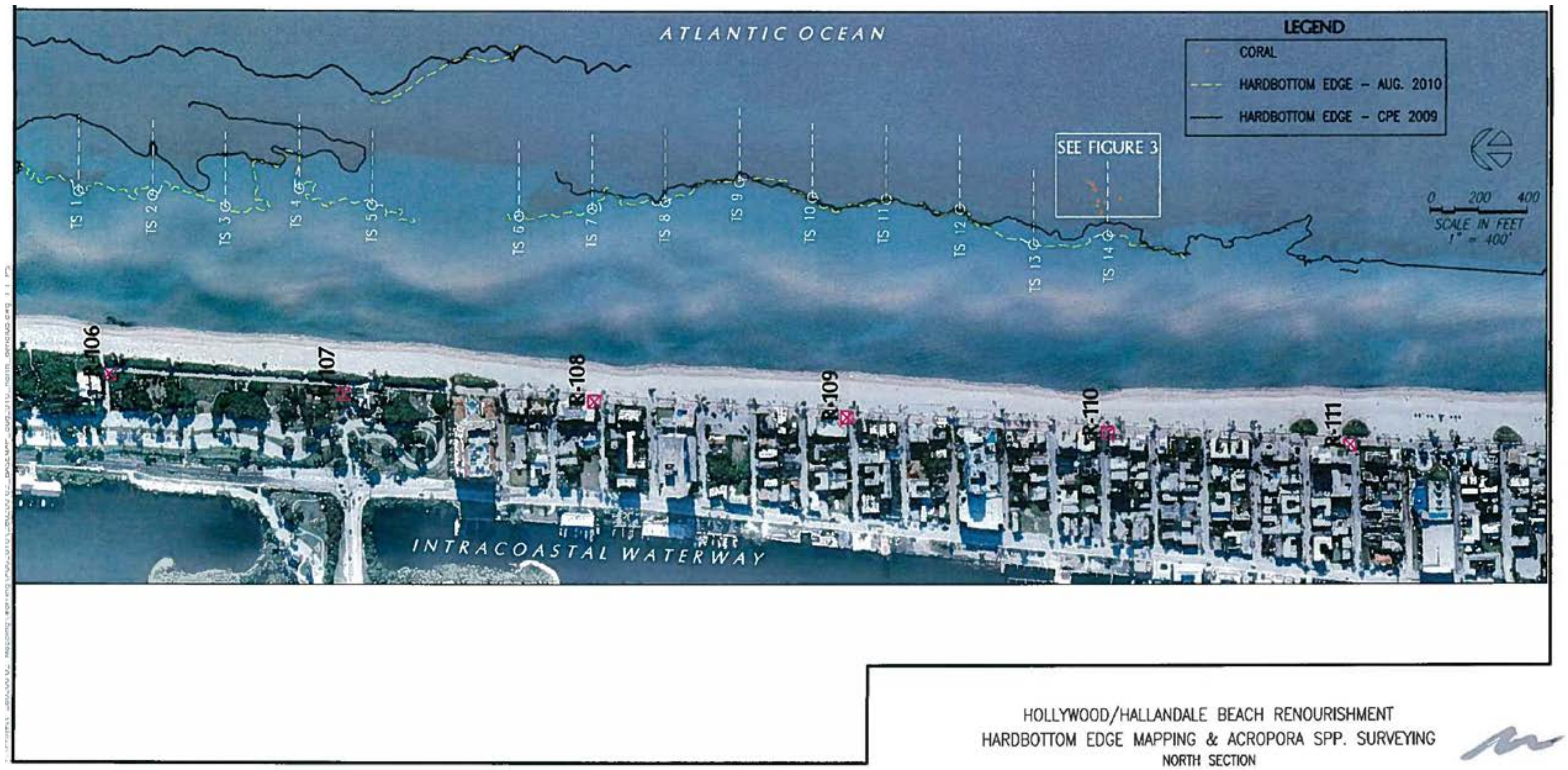

Figure 47. Staghorn coral colonies located in Broward Segment 3, north Section Hollywood/Hallandale Beach (NMFS 2011b).

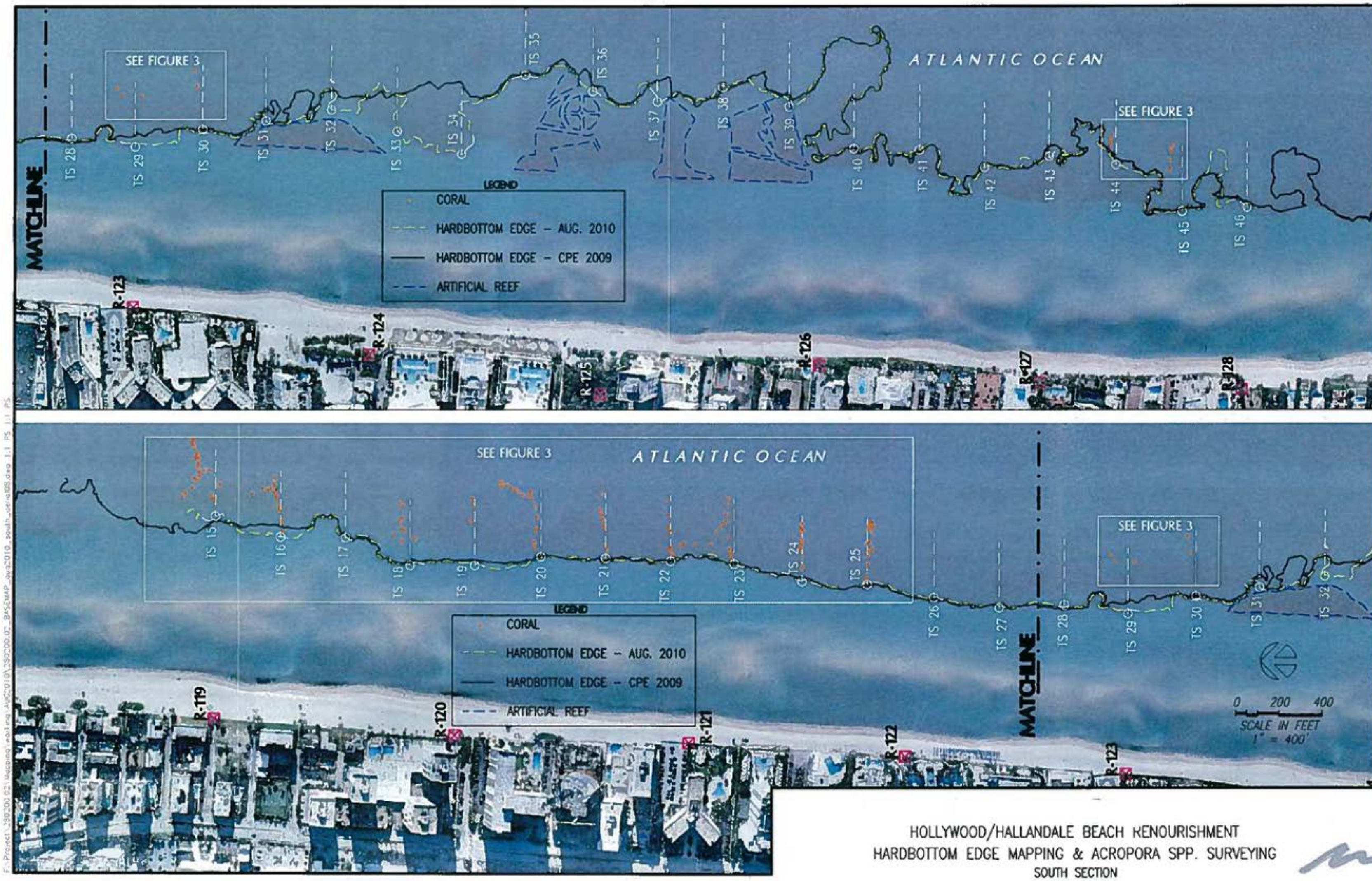

Figure 48. Staghorn coral colonies located in Broward Segment 3, South Section Hollywood/Hallandale Beach (NMFS 2011b).

6.3.1 Beach Nourishment

As discussed above, this Opinion only covers beach nourishment projects within the range of ESA-listed corals for beaches that have been previously nourished and the entire limits of the previously authorized/permitted beach fill template has been constructed, excluding the US Caribbean. This Opinion does not cover projects that place sand on coral hardbottom in areas not previously constructed. Past beach nourishment projects for which information was available on the presence of corals ranged from Jupiter Carlin in Palm Beach County to the north down to Miami Beach in Miami Dade County and occurred between 2008 and 2017 (Table 47).

Table 47 contains the best available information to analyze the expected level of take of ESA-listed corals from beach nourishment; however, detailed survey information is not available for every beach nourishment site that will be covered under this Opinion. We accordingly consider the available information representative of conditions in beach nourishment locations in similar areas nearby for purposes of this analysis. However, any project involving beach nourishment within the range of ESA-listed corals will require project-specific surveys, and project-specific review by NMFS, as described in the PDCs (Appendix C), to ensure that effects are consistent with those analyzed in this Opinion.

Of the information available on the presence of coral colonies near beaches where beach nourishment is expected to occur under this Opinion (Table Table 47), only Broward County Segment 2 and Segment 3 were identified as having nearby ESA-listed corals (Table 47) and only Segment 2 was identified as having ESA-listed corals within 500 ft of the ETOF that will be surveyed prior to a beach nourishment project covered under this Opinion (NMFS 2011b), shown in Figure 46 -Figure 48. All identified ESA-listed corals at these locations were either staghorn or elkhorn corals. No ESA-listed species other than staghorn and elkhorn corals were identified as present in any of the previous beach nourishment projects, and we believe these other ESA-listed coral species are unlikely to occur within 500 ft of the ETOF of beach nourishment projects covered under this Opinion, because they are unlikely to occur in such close proximity to shore (see Section 4.1.6 of this Opinion).

For Broward County Section 3, the maps of the ETOF (Figure 46), hardbottom edge (NMFS 2011b), Figure 46 -Figure 48), and presence of staghorn coral colonies (Figure 47 and Figure 48), show that coral hardbottom and all the staghorn coral colonies are located farther than 500 ft beyond the ETOF in the Hollywood and Hallandale sections of Segment 3. No information is available for the other portions of Segment 3; therefore, we expect that colonies in this area will located over 500 ft beyond the ETOF similar to the conditions present off the Hollywood and Hallandale portions of Segment 3 (no coral hardbottom or staghorn colonies present within 500 ft of the ETOF.

In Broward County Segment 2, there were up to 13,153 colonies of staghorn coral (using the highest number in the abundance range categories in Table 47 -Table 49 and 1 colony of elkhorn coral identified within 500 ft of the hardbottom edge (Coastal Planning & Engineering 2013). From the maps Figure 46 -Figure 48, and using the highest number in the abundance range categories, we estimate 5,391 staghorn coral colonies occur within 500 ft of the ETOF and 1,232 staghorn coral colonies occur within 100 ft of the ETOF. Based on past experience with Section 7 consultations on sedimentation impacts on from beach placement, the implementation of the

PDCs, and the use of beach quality sand to nourish in these locations, we expect that any harmful sedimentation impacts outside the ETOF would be limited to within 100 ft of the ETOF. Therefore, we base our analysis on the scenario of relocating the 1,232 colonies of staghorn coral and 1 colony of elkhorn coral within 100 ft of the ETOF at Broward Section 2. However,this may be an over estimate since sedimentation effects to corals adjacent to the ETOF should only occur in limited instances, based on site specific conditions.

Available information indicates that ESA-listed corals have only been found in beach nourishment locations within 500 ft of the ETOF in Broward County Sections 2 and Section 3; the majority of the beach nourishment projects previously completed and discussed in Table 47 did not have corals adjacent to the beach nourishment areas covered under this Opinion. Therefore, since we have no information to suggest that ESA-listed corals will be present in other beach nourishment locations used as part of the proposed action, and the coral relocation take estimated for the 1 project that reported corals near the ETOF may be an overestimate since corals that would need to be relocated would be limited to those that recruited into previously nourished beach areas within the range of corals and that elkhorn corals would be limited to only portions of Broward and Miami-Dade County, Florida, we do not calculate additional effects to corals from beach nourishment. We assume that the effects calculated for Broward County Section 2 will include all relocation necessary for all beach nourishment projects. Since the data reviewed occurred over a 10 year period (2008-2017), we assume that the estimated take will include the potential need to relocate these coral species during any consecutive 10-year period under this Opinion.

6.3.2 Pipelines

As discussed, the PDCs limit pipeline placement under this Opinion to existing pipeline corridors within the range of ESA-listed corals. Corals that may have been located in these routinely used corridors were relocated during past projects that used the pipeline corridor, which are limited to South Florida and do not include the U.S. Caribbean. We reviewed available information on completed ESA Section 7 consultations for pipelines and only found 1 project that required relocation of coral colonies from a pipeline corridor, at the Miami Beach north pipeline corridor in 2012 (NMFS 2011a). That Opinion only identified the presence of staghorn corals, and authorized relocation of 68 colonies. However, that Opinion was completed before the listing of pillar, rough cactus, lobed star, mountainous star, and boulder star coral. We accordingly believe that other ESA-listed coral species may be present in pipeline corridors within the range of ESA-listed corals, including boulder star, elkhorn, lobed star, and mountainous star coral noting that we do not expect recruitment of pillar or rough star cactus due to their low density in the project area, as discussed at the beginning of Section 6.3 of this Opinion. NOAA-compiled data from benthic surveys conducted in Florida in 2018 (NOAA, National Centers for Coastal Ocean Science (NCCOS), unpublished data) indicate that average density is similarly widely distributed for staghorn coral, elkhorn coral, lobed star coral, and boulder star coral (Table 48). Therefore, we assume that other corals may be present in pipeline corridors covered under this Opinion and that the number of corals that may be in these pipeline corridors will be in similar densities to those identified in the benthic survey. Using this data, we estimated the number of corals that may need to be relocated by determining the percent difference of the density of corals observed in the Florida 2018 (NOAA, unpublished data) comparing elkhorn, lobed star, and boulder star corals to staghorn corals since we know the number of staghorn corals observed in the pipeline corridor project at Miami Beach discussed above. The density percent difference was then applied to determine the difference in the number of corals relative to the 68 staghorn corals observed in Miami Beach to estimate the number of corals that may be found in other pipeline projects covered under this Opinion. For example, the density of elkhorn coral was 0.001 and the density of staghorn corals was 0.039, which means that elkhorn corals are 7.44% less likely to be found than staghorn corals. Therefore, if 68 staghorn corals are expected to be present, only 1.74 elkhorn corals would be expected to be present (i.e., 7.44% of 68 corals).

We assume the estimated number of corals that will be relocated in pipeline corridors for projects covered under this Opinion will include all corals that will need to be relocated during a 10 year period of this Opinion, based on the single project that has reported the removal of corals for a pipeline in the action area during the 10 years of pipeline project data. Projects involving pipeline use in the range of ESA-listed coral and under this Opinion are expected to be limited. The USACE reports that only 1-2 pipeline projects may occur in a given year and beach nourishment projects in South Florida are relying more on upland sources of sand for beach nourishment and less on pipeline delivery from offshore sources due to diminished offshore supply of sand. The number of corals that may need to be relocated is also assumed to be minimal, since pipeline corridors are reused over time and most of the ESA-listed coral colonies that could occur within the corridors will have been previously removed and will be slow to return due to low recruitment rates of ESA-listed coral species.

Table 48. Estimated Corals to be Relocated for Pipeline Projects

Species	**Florida Coral Density (colonies per m^2)[80]**	**Percent Difference of Florida Coral Density than Staghorn Coral**	**Relocation Estimate[81]**
Staghorn coral	0.039	0	68.00
Elkhorn coral	0.001	-97.44%	1.74
Lobed star coral	0.029	-25.64%	50.56
Mountainous star coral	0.092	135.90%	160.41
Boulder star coral	0.042	7.69%	73.23

6.3.3 Total Coral Relocations

While relocation is a minimization measure to reduce the risk of lethal take of ESA-listed corals, relocation is itself take and can also result in mortality. Coral transplantation can successfully relocate colonies that would likely suffer partial or total morality if not moved. Provided that colonies are handled with skill, are reattached properly, and the environmental factors at the reattachment site are conducive to their growth (e.g., water quality, substrate type), many different species of coral have been shown to survive transplantation well (Becker and Mueller 2001; Guzmán 1991; Hudson and Diaz 1988; Lindahl 2003). Schopmeyer et al. (2017) documented an 85% survival rate in staghorn coral fragments 12 months after transplantation

[80] Florida coral benthic survey used to determine density of corals in the area (NOAA, unpublished data)
[81] (Species Percent Density from staghorn corals x 68 [number of staghorn corals observed in 2012 pipeline survey] DERM. 2012. Relocation of Acroporid corals from the Miami Beach-North pipeline corridor in association with the Miami-Dade test beach nourishment. Miami-Dade County Department of Environmental Resource Management, FDEP JCP Permit File #0295427-001JC, Miami, Florida.

from multiple coral nurseries across Florida and in Puerto Rico. Based on previous transplant survival rates and the protective measures in the Coral PDCs Appendix C, we conclude that low post-transplantation mortality is likely to occur. Even though transplantation of corals outside the project area involves directed take, it reduces the level of lethal take by relocating colonies to an area where they will have a high likelihood of continued survival. The PDCs of this Opinion require that corals be relocated by someone trained to perform this task and be handled and reattached at the new location in ways that optimize the success of transplanting. For this Opinion, we will assume a 15% morality rate of relocated coral colonies, as discussed above. The colony survival rate is defined as having at least 75% living tissue on the transplanted colonies, according to the Coral PDCs in Appendix C. Surveys will be completed at 1 week, 1 month, 3 months, 6 months, and 1 year to ensure that survival rates are consistent with the analysis in this Opinion. The Coral PDCs also provide the option to coordinate with local coral nurseries to see if corals can be given to the nursery instead of outplanting. Corals given to coral nurseries can then be grown and divided to outplant at multiple locations, which helps to increase the density of corals in the area. Coral provided to a nursery do not need to be monitored for transplantation success under this Opinion. For the purposes of this Opinion, we still consider those provided to a nursery to have 15% mortality for the purposes of the Jeopardy Analysis, but assume this may be an overestimate since coral nursery staff are trained to handle these species and monitor them frequently.

To calculate the lethal and nonlethal take of ESA-listed corals estimated to occur under projects covered under this Opinion, we combine the estimated number of corals calculated for beach nourishment projects and pipeline projects in the prior section and apply the estimated 15% relocation mortality rate in Table 49, which will be analyzed further in the Jeopardy Analysis in Section 8 of this Opinion.

Table 49. Estimated Nonlethal and Lethal Take of ESA-listed Corals from Beach Nourishment Projects and Associated Pipeline Corridors During 10-years of Activities Covered Under this Opinion.

Column Number	Calculations Used to Estimate the Number of Colonies	Staghorn Coral Colonies	Elkhorn Coral Colonies	Lobed Star Coral Colonies	Mountainous Star Coral Colonies	Boulder Star Coral Colonies
1	Beach Nourishment Relocations from Table 47	1,232	1	0	0	0
2	Pipeline Relocations from Table 48	68.00	1.74	50.56	160.41	73.23
3	Total Relocation Estimate (Column 2 + 3) (Rounded Up)	1,300	3	51	161	74
4	Estimated Lethal Take (Column 3 x 15% Mortality)	195.00	1	7.58	24.06	10.98
5	Estimate Lethal Take (Column 4, Rounded Up)	195	1	8	25	11
6	Estimated Non-lethal Take (Column 3-5)	1,105	2	43	136	63

7 CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area considered in this Biological Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

Human-induced mortality and/or injury of ESA-listed sea turtles, sturgeon, Nassau grouper, elasmobranchs, and whales occurring in the action area are reasonably certain to occur in the future. The sources of those effects include vessel interactions, marine debris, pollution, global climate change, and coastal development. While the combination of these activities may prevent or slow the recovery of these species, the magnitude of these effects is currently unknown.

7.1 Vessel Interactions

ESA-listed species stranding reports indicate that sea turtles, sturgeon, giant manta ray, and whales are struck and killed by vessels within the action area each year, as discussed in the Status of the Species in Section 4 and in the Baseline in Section 5.4 of this Opinion as part of past and ongoing actions. Such collisions are reasonably certain to continue into the future through the continued use of vessels in the action area.

7.2 Pollution

Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them on ESA-listed species, as discussed in the Status of the Species in Section 4 of this Opinion and in the Baseline in Section 5.6 of this Opinion as part of past and ongoing actions. Marine debris (e.g., discarded fishing line or lines from boats) and vertical lines in the water are known to entangle ESA-listed species resulting in injury or death and can entangle ESA-listed corals resulted in damage or loss of corals. In addition, ESA-listed species commonly ingest plastic or mistake debris for food. Excessive turbidity due to coastal development and/or construction sites could influence the ability for these species to use highly turbid areas or can bury non-mobile species such as Johnson's seagrass and ESA-listed corals. However, the level of future impacts cannot be projected.

7.3 Sound

In-water noise is reasonably certain to continue in the future and effect ESA-listed species, as discussed in the Status of the Species in Section 4 of this Opinion and in the Baseline in Section 5.4 of this Opinion as part of the past and ongoing actions. The noise level in the ocean is thought to be increasing at a substantial rate due to increases in shipping and other activities, including seismic exploration, offshore drilling, and sonar used by military and research vessels. Concerns about noise in the action area of this consultation include increasing noise due to increasing commercial shipping, recreational vessels, and geophysical surveys.

7.4 Global Climate Change

Global climate change is likely adversely affecting ESA-listed species in the action area. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events and fluctuation of precipitation levels, and change in air and water temperatures. The effects on ESA-listed species are unknown at this time. There are multiple hypothesized effects to ESA-listed species, including changes in their range and distribution as well as prey distribution and/or abundance due to water temperature changes. Ocean acidification may also negatively affect marine life, particularly organisms with calcium carbonate shells that serve as important prey items for many species. For example, global climate change may affect sea turtles by changing their reproductive behavior including earlier onset of nesting, shorter intervals between nesting, and a decrease in the length of nesting season. Sea level rise may also reduce the amount of nesting beach available and changes in air temperature may also affect the sex ratio of sea turtle hatchlings. A decline in sea turtle reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of sea turtles in the Atlantic. Changing ocean temperatures and acidification will also affect non-mobile species such as ESA-listed coral and Johnson's seagrass.

Sea levels and water temperatures are expected to rise, and levels of precipitation are likely to fluctuate. Drought and inter- and intra-state water allocations and their associated impacts to ESA-listed species will continue and may intensify. For example, a rise in sea level may drive the salt wedge upriver on river systems inhabited by Atlantic and shortnose sturgeon, potentially constricting their habitat. NMFS will continue to work with states to implement ESA Section 6 agreements, and with researchers holding Section 10 permits, to enhance programs to quantify and mitigate these takes and effects.

7.5 Coastal Development

Within the action area, beachfront development, lighting, and beach erosion potentially reduce or degrade sea turtle nesting habitats or interfere with hatchlings movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. Coastal counties are presently adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting. Some of these measures were drafted in response to lawsuits brought against the counties by concerned citizens who charged the counties with failing to uphold the ESA by allowing unregulated beach lighting that results in takes of hatchlings.

7.6 Summary

Beyond the threats noted above, NMFS is not aware of any proposed or anticipated changes in other human-related actions (e.g., poaching, habitat degradation) or natural conditions (e.g., overabundance of land or sea predators, changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on ESA-listed species covered by this Opinion.

8 Jeopardy Analysis

The analyses conducted in the previous sections of this Opinion provide the basis on which we determine whether the proposed action would be likely to jeopardize the continued existence of the following species: green sea turtle (NA & SA DPSs), Kemp's ridley sea turtle, leatherback sea turtles, loggerhead sea turtle (NWA DPS), smalltooth sawfish (U.S. DPS), shortnose sturgeon, Atlantic sturgeon (New York Bight, Chesapeake Bay, Gulf of Maine, Carolina, and SA DPS), giant manta ray, Johnson's seagrass, elkhorn coral, staghorn coral, boulder star coral, mountainous star coral, and lobed star coral.

In the effects of the action (Section 6 of this Opinion), we outlined how the proposed action would affect these species at the individual level and the magnitude of those effects based on the best available data. Now, we assess each of these species' response to the effects of the proposed action in terms of overall population effects and whether those effects when considered in the context of the status of the species, the environmental baseline, and the cumulative effects, are likely to jeopardize their continued existence in the wild.

It is the responsibility of the action agency to "insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered species or threatened species…" (ESA Section 7(a)(2)). Action agencies must consult with and seek assistance from the NMFS to meet this responsibility. NMFS must ultimately determine in a Biological Opinion whether the action jeopardizes listed species. To *jeopardize the continued existence of* is defined as "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Thus, in making this determination for each species, we must look at whether the proposed actions directly or indirectly reduce the reproduction, numbers, or distribution of a listed species. Then, if there is a reduction in 1 or more of these elements, we evaluate whether it would be expected to cause an appreciable reduction in the likelihood of both the survival and the recovery of the species.

The actions considered in this Opinion that we believe are likely to result in take are primarily actions that have been occurring in the action area for many decades and are expected to continue to occur. This Opinion considers the cumulative effects of the continued maintenance of these activities that were previously considered under multiple Opinions, and includes, via the PDCs, minimization measures that are expected to reduce both the likelihood and impact of take from the proposed action. This Opinion will remain in effect until the reinitiation triggers outlined in Section 12 are met and accordingly does not have a specific length of time in which activities will be covered. Therefore, we considered it appropriate to consider the effects of take to the species analyzed in this section based on the number of take estimated over any consecutive 3-year period for species with take estimated to occur annually (as described in Section 6) and over longer periods of time for those species that are not estimated to have take annually (i.e. ESA-listed corals and smalltooth sawfish). The loss of individuals over the period of time specified in the analysis for each species is a reduction in numbers and the cumulative effects are analyzed in this section. We acknowledge that the loss of these species over the length of the Opinion will result in loss over multiple consecutive periods of the time period specified (e.g., over multiple consecutive 3-year periods). However, since these activities are a largely a continuation of activities that are already occurring, we believe that the take considered in this Opinion will result in impacts that are similar to those that have been occurring to the populations of these species over decades, while these maintenance activities have been ongoing in the action area. Therefore, and given our experience monitoring ITSs based on consecutive time periods, we believe that it is appropriate to monitor and analyze take and the impact of take on this basis. We believe that basing the jeopardy analysis on consecutive time periods accurately accounts for impacts resulting from continued maintenance of activities covered under this Opinion that may, directly or indirectly, reduce appreciably the likelihood of survival or recovery of the listed species in the wild, even when considered over the life of the Opinion. Since work will occur throughout the expansive action area, the reduction in numbers of each species will also be distributed as discussed for each species below.

8.1 Green Sea Turtle (NA and SA DPSs)

Mixed-stock analyses of foraging grounds show that green sea turtles from multiple nesting beaches commonly mix at feeding areas across the Caribbean and Gulf of Mexico, with higher contributions from nearby large nesting sites and some contribution estimated from nesting populations outside the DPS (Bass et al. 1998; Bass and Witzell 2000; Bjorndal and Bolten 2008; Bolker et al. 2007). In other words, the proportion of animals on the foraging grounds from a given nesting beach is proportional to the overall importance of that nesting beach to the entire DPS. For example, Tortuguero, Costa Rica, is the largest nesting beach in the North Atlantic DPS and the number of animals from that nesting beach on foraging grounds were higher than from any other nesting beach. More specifically, Lahanas et al. (1998) showed that juvenile green sea turtles in the Bahamas originate mainly from the western Caribbean (Tortuguero, Costa Rica) (79.5%) (North Atlantic DPS) but that a significant proportion may be coming from the eastern Caribbean (Aves Island/Suriname; 12.9%) (SA DPS).

Flipper tagging studies provide additional information on the co-mingling of turtles from the North Atlantic DPS and South Atlantic DPS. Flipper tagging studies on foraging grounds and/or nesting beaches have been conducted in Bermuda (Meylan et al. 2011), Costa Rica (Troëng and Rankin 2005), Cuba (Moncada et al. 2006), Florida (Johnson and Ehrhart 1996; Kubis et al. 2009), Mexico (Zurita et al. 2003; Zurita et al. 1994), Panama (Meylan et al. 2011), Puerto Rico (Collazo et al. 1992; Patrício et al. 2011), and Texas (Shaver 1994; Shaver 2002). Nesters have been satellite tracked from Florida, Cuba, Cayman Islands, Mexico, and Costa Rica. Troëng and Rankin (2005) report that while there is some crossover of adult female nesters from North Atlantic DPS into the South Atlantic DPS, particularly in the equatorial region where the DPS boundaries are in closer proximity to each other, North Atlantic DPS nesters primarily use the foraging grounds within the North Atlantic DPS.

Since this action area includes projects from the southeast continental United States as well as projects in the U.S. Caribbean (i.e., Puerto Rico and U.S. Virgin Islands), green sea turtles from both the North Atlantic and South Atlantic DPSs can be found within the action area. While there are currently no in-depth studies available to determine the percent of North Atlantic and South Atlantic DPS individuals in any given location, an analysis of cold-stunned green turtles in St. Joseph Bay, Florida (northern Gulf of Mexico) found approximately 4% of individuals came from nesting stocks in the SA DPS. On the Atlantic coast of Florida, a study on the foraging grounds off Hutchinson Island found that approximately 5% of the turtles sampled came from the SA DPS (Bass and Witzell 2000). All of the individuals in both studies were benthic juveniles.

As discussed in Section 6 of this Opinion, the vast majority of the projects that will result in take of sea turtles will occur in the continental United States and the only take in the U.S. Caribbean will be limited to maintenance hopper dredging without relocation trawling in up to 3 locations every 5 years, which is not expected to result in a statistically significant difference in the composition of North Atlantic verses South Atlantic DPSs takes annually. The information available on the composition of green sea turtle DPSs suggests that the vast majority of the anticipated captures in the South Atlantic region of the continental United States are likely to come from the North Atlantic DPS. To analyze effects in a precautionary manner, we will conduct 2 jeopardy analyses assuming 5% could come from the SA DPS and 95% will be from the NA DPS.

Based on the total estimated take provided in Table 46, we applied the percent of take that will be from each DPS to determine the number of nonlethal and lethal take of green sea turtles in Table 50. We also estimate that 3 gravid females will be relocated resulting in a lost clutch. Since all relocation trawling will occur outside of the U.S. Caribbean, we assume any gravid females encountered during relocation trawling would be NA DPS and therefore the lost clutches will all be of the NA DPS. The estimated 62 lethal green sea turtle take include 1.83 sea turtles estimated to be captured during any consecutive three year period that will not be able to be identified when retrieved dismembered.

Table 50. 3-Year Estimated Take of Green Sea Turtle NA and SA DPS

	Non-Lethal Take Observed	Lethal Take Observed	Lethal Take Unobserved
Total take from Table 46	781	62	62
NA DPS (95% of total take)	741.95	58.9	58.9
NA DPS (rounded up)	742	59	59
SA DPS (5% of total take)	39.05	3.1	3.1
SA DPS (rounded up)	40	4	4

8.1.1 NA DPS

The proposed actions covered under this Opinion may result in the following take of NA DPS green sea turtles over any consecutive 3-year period (Table 50):

- Non-lethal take= 742 observed.
- Lethal take= 59 observed and 59 unobserved for a total of 118 lethal take. The estimated 118 lethal NA DPS green sea turtle take includes 1.83 green sea turtles estimated to be captured each year that will not be able to be identified when retrieved dismembered or were unobserved.
- 3 lost egg clutches.

8.1.1.1 Survival

The nonlethal capture of 742 NA DPS green sea turtle over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. Nonlethal take will not result in a reduction in numbers of the species. Additionally, the individuals identified as nonlethal take are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, thus no change in the distribution of NA DPS green sea turtles is anticipated.

The lethal take of 118 green sea turtles from the NA DPS over any consecutive 3-year period of the Opinion is a reduction in numbers. These lethal takes could also result in a potential reduction in future reproduction, assuming some individuals would be female and would have survived to reproduce in the future. For example, an adult green sea turtle can lay 3-4 clutches of eggs every 2-4 years, with approximately 110-115 eggs/nest, of which a small percentage is expected to survive to sexual maturity. For this Opinion, we also specifically considered the likelihood of post-interaction mortality to female sea turtles captured during relocation trawling and the likelihood of those females to each loose a single clutch of eggs (3 total for green sea turtle NA DPS per 3-year period) due to the stress of being captured in a relocation trawling net while gravid (Section 6.1.4.1.2 of this Opinion). The anticipated lethal take of 118 NA DPS green sea turtles from dredging and relocation trawling, including captures of gravid females in the summer, are expected to occur at discrete project locations within the SARBO action area (i.e., where work will occur) and green sea turtles in the NA DPS generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In the *Status of Species* section of this Opinion (Section 4.1.1.2 of this Opinion), we presented the status of the DPS, outlined threats, and discussed information on estimates of the number of nesting females and nesting trends at primary nesting beaches. In the Environmental Baseline, this Opinion outlined the past and present impacts of all state, federal, or private actions and other human activities in or having effects in the action area that have affected and continue to affect this DPS. The Cumulative Effects section of this Opinion discussed the effects of future state, tribal, local, or private actions that are reasonably certain to occur within the SARBO action area.

Seminoff et al. (2015) estimated that there are greater than 167,000 nesting green sea turtle females in the NA DPS. The nesting at Tortuguero, Costa Rica, accounts for approximately 79% of that estimate (approximately 131,000 nesters), with Quintana Roo, Mexico, (approximately 18,250 nesters; 11%), and Florida, USA (approximately 8,400 nesters; 5%), also accounting for a large portion of the overall nesting (Seminoff et al. 2015). At Tortuguero, Costa Rica, the number of nests laid per year from 1999 to 2010 increased, despite substantial human impacts to the population at the nesting beach and at foraging areas (Campell and Lagueux 2005; Troëng 1998; Troëng and Rankin 2005). Nesting locations in Mexico along the Yucatan Peninsula also indicate the number of nests laid each year has deposited, but by 2000 this increased to over

1,500 nests/year (NMFS and USFWS 2007a). By 2012, more than 26,000 nests were counted in Quintana Roo (J. Zurita, El Centro De Investigaciones De Quintana Roo, unpublished data, 2013, in Seminoff et al. (2015)). In Florida, most nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al. 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). As described in Section 4.1.1.2 of this Opinion, nesting has increased substantially over the last 20 years and peaked in 2017 with 53,102 nests statewide in Florida, though the number of nests dropped again in 2018 as part of the regular biennial fluctuation (Figure 25).

In summary, green sea turtle nesting at the primary nesting beaches within the range of the NA DPS has been increasing over the past 2 decades, against the background of the past and ongoing human and natural factors (i.e., the environmental baseline) that have contributed to the current status of the species. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information for NA DPS green sea turtles is increasing, we believe the potential lethal take of 118 green sea turtles and 3 egg clutches from the NA DPS, and 742 nonlethal takes, over any consecutive 3-year period of the Opinion will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed actions, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed actions covered under this Opinion are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the green sea turtle NA DPS in the wild.

8.1.1.2 Recovery

The NA DPS of green sea turtles does not have a separate recovery plan at this time. However, an Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) does exist. Since the animals within the NA DPS all occur in the Atlantic Ocean and would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide until a new plan, specific to the NA DPS, is developed. The Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

> Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.
>
> Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

According to data collected from Florida's index nesting beach survey from 1989-2018, green sea turtle nest counts across Florida have continued to have biennial variability with the highest number of nests reported increasing dramatically since 2013 (Figure 25). Based on the total nests reported in Figure 25, the 10 year nesting average is 13,063, indicating that the first listed recovery objective is currently being met. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting, however, it is likely that numbers on foraging grounds have also increased, consistent with the criteria of the second listed recovery objective.

The potential lethal take of 118 green sea turtles and 3 egg clutches from the NA DPS over any consecutive 3-year period of the Opinion will result in a reduction in numbers when it occurs, but it is unlikely to have any detectable influence on the recovery objectives and trends noted above, even when considered in the context of the of the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. Similarly, we do not expect the nonlethal take of 742 NA DPS green sea turtles to have any detectable influence on the recovery objectives. Thus, the proposed actions covered under this Opinion will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of NA DPS green sea turtles' recovery in the wild.

8.1.1.3 Conclusion

The nonlethal take of 742 and lethal take of 118 green sea turtles plus the loss of 3 green sea turtle egg clutches over any consecutive 3-year period from the NA DPS associated with the proposed actions covered under this Opinion are not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the NA DPS of green sea turtles in the wild.

8.1.2 SA DPS

Until we have an analysis with a larger sample size across more of the Atlantic by the NMFS Southeast Fisheries Science Center genetics lab and because we do not anticipate any impacts to nesters or hatchlings, we use 5% as the best estimate of the percent of SA DPS individuals that could be taken by a project in waters of the Atlantic Ocean, including projects covered under this Opinion in the U.S. Caribbean.

The proposed actions covered under this Opinion may result in the following take of SA DPS green sea turtles over any consecutive 3-year period (Table 46):

- Non-lethal take= 40 observed
- Lethal take= 4 observed and 4 unobserved for a total of 8 lethal take.

8.1.2.1 Survival

The nonlethal capture of 40 SA DPS green sea turtle over any 3 consecutive year period, is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Nonlethal take will not result in a reduction in numbers of the species. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of SA DPS green sea turtles is anticipated.

The lethal take of 8 green sea turtle from the SA DPS over any consecutive 3-year period of the Opinion is a reduction in numbers. As discussed above, lethal interactions would also result in a potential reduction in future reproduction, assuming the individual would be female and would have survived otherwise to reproduce. For this Opinion, we considered the likelihood of post-interaction mortality to female sea turtles captured during relocation trawling and the likelihood of those females to lose a clutch of eggs due to the stress of being captured in a relocation trawling net while gravid (Section 6.1.4.1.2 of this Opinion) and do not anticipate any lost clutch of eggs from the SA DPS since relocation trawling is not covered under this Opinion in the U.S. Caribbean, where SA DPS females would be nesting.

Whether the potential reduction in numbers of up to 8 green sea turtles would appreciably reduce the likelihood of survival of green sea turtles from the SA DPS depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends. In Section 4.1.1.2 of this Opinion, we reviewed the status of the species in terms of nesting and female population trends and several recent assessments based on population modeling (e.g., (Seminoff et al. 2015). In Section 5 of this Opinion, we evaluated the Environmental Baseline, including known sources of mortality affecting sea turtle populations in the action area. The SA DPS is large, estimated at over 63,000 nesting females, but data availability is poor with 37 of the 51 identified nesting sites not having sufficient data to estimate number of nesters or trends (Seminoff et al. 2015). While the lack of data was a concern due to increased uncertainty, the overall trend of the SA DPS was not considered to be a major concern as some of the largest nesting beaches such as Ascension Island and Aves Island in Venezuela and Galibi in Suriname appear to be increasing with others (Trindade, Brazil; Atol das Rocas, Brazil; Poilão and the rest of Guinea-Bissau) appearing to be stable. In the U.S., nesting of green sea turtles occurs in the SA DPS on beaches of the U.S. Virgin Islands, primarily on Buck Island and Sandy Beach, St. Croix, although there are not enough data to establish a trend. We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of green sea turtles from the SA DPS in the wild. Although the potential mortality of up to 8 sea turtles from this DPS over any 3 consecutive year period may occur as a result of the proposed action and would result in a reduction in absolute population numbers, the population of green sea turtles in the SA DPS would not be appreciably affected. Likewise, the reduction in reproduction that could occur due to lethal take of the individuals would not appreciably affect reproduction output in the South Atlantic.

8.1.2.2 Recovery

Like the NA DPS, the SA DPS of green sea turtles does not have a separate recovery plan in place at this time. However, an Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991) does exist. Since the animals within the SA DPS all occur in the Atlantic Ocean and would have been subject to the recovery actions described in that plan, we believe it is appropriate to continue using that Recovery Plan as a guide until a new plan, specific to the SA DPS, is developed. In our analysis for the NA DPS, we stated that the Atlantic Recovery Plan lists the following relevant recovery objectives over a period of 25 continuous years:

> Objective: The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years.
>
> Objective: A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

The nesting recovery objective is specific to the NA DPS, but demonstrates the importance of increases in nesting to recovery. As previously stated, nesting at the primary SA DPS nesting beaches has been increasing over the past 3 decades. There are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds. Given the clear increases in nesting and in-water abundance, however, it is likely that numbers on foraging grounds have increased.

The potential for 8 lethal green sea turtle take from the SA DPS over any consecutive 3-year period of the Opinion will result in a reduction in numbers when it occurs, but it is unlikely to have any detectable influence on the trends noted above, even when considered in context with the Status of the Species, the Environmental Baseline, and Cumulative Effects discussed in this Opinion. Similarly, we do not expect the nonlethal take of 40 SA DPS green sea turtles to have any detectable influence on the recovery objectives. Thus, the proposed action will not impede achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of the SA DPS of green sea turtles' recovery in the wild.

8.1.2.3 Conclusion

The potential for 40 nonlethal and 8 lethal green sea turtle take over any consecutive 3-year period from the SA DPS associated with the proposed actions covered under this Opinion is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of the SA DPS of green sea turtles in the wild.

8.2 Kemp's ridley Sea Turtle

The proposed actions covered under this Opinion may result in the following take of Kemp's ridley sea turtles over any consecutive 3-year period (Table 46):

- Non-lethal take= 1,340 observed.
- Lethal take= 58 observed and 58 unobserved for a total of 116 lethal take. The estimated 116 lethal Kemp's ridley sea turtle take include 1.73 sea turtles estimated to be captured each year that will not be able to be identified when retrieved dismembered.
- 1 lost egg clutches.

8.2.1 Survival

The nonlethal capture of 1,340 Kemp's ridley sea turtle over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of Kemp's ridley sea turtles is anticipated.

The potential lethal take of up to 116 Kemp's ridley sea turtle over any consecutive 3-year period for projects covered under this Opinion is a reduction in numbers. The Turtle Expert Working Group (1998) estimates age at maturity from 7-15 years. Females return to their nesting beach about every 2 years (Turtle Expert Working Group 1998). The mean clutch size for Kemp's ridley sea turtle is 100 eggs/nest, with an average of 2.5 nests/female/season. Lethal takes could also result in a potential reduction in future reproduction, assuming at least 1of these individuals would be female and would have survived to reproduce in the future. The loss of 116 Kemp's ridley sea turtles over consecutive 3 year periods could preclude the production of thousands of eggs and hatchlings (outside the action area), of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of any females would eliminate their contribution to future generations, and result in a reduction in sea turtle reproduction. For this Opinion, we considered the likelihood of post-interaction mortality to female sea turtles captured during relocation trawling and the likelihood of those females to each loose a single clutch of eggs (1 total for Kemp's ridley sea turtle per 3-year period) due to the stress of being captured in a relocation trawling net while gravid (Section 6.1.4.1.2 of this Opinion). The anticipated lethal take of Kemp's ridley green sea turtles from dredging and relocation trawling, including gravid females during the summer, are expected to occur at discrete project locations within the SARBO action area (i.e., where work will occur) and Kemp's ridley sea turtles generally have large ranges; thus, no reduction in the distribution is expected from the lethal take of these individuals.

In the absence of any total population estimates for Kemp's ridley sea turtle, nesting trends are the best proxy for estimating population changes. Since 2008, the biennial nesting data for Mexican Beaches has fluctuated, but remains higher than all nesting years since 2005 (Figure 26). A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 42 in 2004, to a record high of 353 nests in 2017 (Section 4.1.1.3.3 of this Opinion). It is worth noting that nesting in Texas has paralleled the trends observed in Mexico, characterized by a significant decline in 2010, followed by a second decline in 2013-2014, but with a rebound in 2015-2017. It is important to remember that with significant inter-annual variation in nesting data, sea turtle population trends necessarily are measured over decades and the long-term trend line better reflects the population increase in Kemp's ridley sea turtles.

Even with reported biennial fluctuations in nesting numbers from Mexican beaches, all years since 2006 have reported over 10,000 nests per year, indicating an increasing population over the previous decades. We believe this long-term increasing trend in nesting is evidence of an increasing population, as well as a population that is maintaining (and potentially increasing) its genetic diversity. We believe these nesting trends are indicative of a species with a high number of sexually mature individuals. Since the abundance trend information is clearly increasing, we believe the potential lethal take of 116 Kemp's ridley sea turtles, the nonlethal take of 1,340 Kemp's ridley sea turtles, and 1 lost egg clutch over any consecutive 3-year period for projects covered under this Opinion will not have any measurable effect on that trend. After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of Kemp's ridley sea turtles in the wild.

8.2.2 Recovery

As to whether the proposed action will appreciably reduce the species' likelihood of recovery, the recovery plan for the Kemp's ridley sea turtle (NMFS et al. 2011) lists the following relevant recovery objective:

> Objective: A population of at least 10,000 nesting females in a season (as measured by clutch frequency/female/season) distributed at the primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) in Mexico is attained. Methodology and capacity to implement and ensure accurate nesting female counts have been developed.

The recovery plan states the average number of nests per female is 2.5; it sets a recovery goal of 10,000 nesting females associated with 25,000 nests. In the last 10 years, 3 of those years reported nests of over 20,000; however, none of the years have reported over 25,000 nests for the Mexican Beaches. It is clear that the population has increased over the last 2 decades and the increase in Kemp's ridley sea turtle nesting is likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of TEDs, reduced trawling effort in Mexico and the United States, and possibly other changes in vital rates (Turtle Expert Working Group 1998; Turtle Expert Working Group 2000). The lethal take of up to 116 Kemp's ridley sea turtles over any consecutive 3-year period for projects covered under this Opinion will result in a reduction in numbers and reproduction, but it is unlikely to have any detectable influence on the nesting trends noted above. Given a nesting population in the thousands, the projected loss is not expected to have any discernable impact to the species recovery objective. Similarly, we do not expect the nonlethal take of 1,340 Kemp's ridley sea turtles to have any detectable influence on the recovery objectives. We accordingly conclude that it is not likely to appreciably reduce the species' likelihood of recovery in the wild.

8.2.3 Conclusion

The lethal take of 116, nonlethal take of 1,340 Kemp's ridley sea turtles, and loss of 1 egg clutch associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of Kemp's ridley sea turtles in the wild.

8.3 Leatherback Sea Turtles

The proposed actions covered under this Opinion may result in the following take of leatherback sea turtles over any consecutive 3-year period (Table 46):

- Non-lethal take= 369 observed.
- Lethal take= 0 observed and 4 unobserved for a total of 4 lethal take. All leatherback will be identifiable, as they will be caught in relocation trawling.
- 6 lost egg clutches.

8.3.1 Survival

The nonlethal capture of 369 leatherback sea turtle over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of leatherback sea turtles is anticipated.

The lethal take of up to 4 leatherback sea turtles over any consecutive 3-year period would reduce the population by that number compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Lethal captures could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and would have survived otherwise to reproduce in the future. For example, an adult female leatherback sea turtle can produce up to 700 eggs or more per nesting season (Schulz 1975). Although a significant portion (up to approximately 30%) of the eggs can be infertile, the annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. For this Opinion, we also considered the likelihood of post-interaction mortality to female sea turtles captured during relocation trawling and the likelihood of those females to each loose a single clutch of eggs (6 total leatherback sea turtles per 3-year period) due to the stress of being captured in a relocation trawling net while gravid (Section 6.1.4.1.2 of this Opinion). While the loss of one clutch would result in the loss of an average of 110 eggs, leatherback sea turtles nest multiple times in a season and therefore would be expected to still lay hundreds of eggs in the same nesting season. The anticipated lethal take of leatherback sea turtles from dredging and relocation trawling, including captures of gravid females in the summer, are expected to occur at discrete project locations within the SARBO action area (i.e., where work will occur) and leatherback sea turtles generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

The Leatherback Turtle Expert Working Group estimated there were between 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic based on 2004 and 2005 nesting count data (Turtle Expert Working Group 2007). The potential loss of up to 4 leatherback sea turtles every 3 years would account for only 0.00005-0.0001% of that North Atlantic population estimate, which is a subset of the listed entity. As discussed in Section 4.1.1.4 of this Opinion, the Northwest Atlantic population has experienced declines, with long-term trends showing as much as a 60% decline in the population, mostly driven by Southern Caribbean/Guianas stock (Northwest Atlantic Leatherback Working Group 2018) however, the loss of up to 4 individuals every 3 years would still represent only a small fraction of one percent of the population. We do not believe these potential loses will have any detectable impact on these population numbers.

Since we anticipate a low number of mortalities (4 per 3 consecutive year period), and the loss of only 6 egg clutches over any consecutive 3-year period for projects covered under this Opinion, we believe the potential mortalities associated with the proposed action will have no detectable effect on current nesting trends. After analyzing the magnitude of the effects of the proposed action, in combinationith the past, present, and future expected impacts to the species discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the leatherback sea turtle in the wild.

Since we do not anticipate the proposed action will have any detectable impact on the population overall, or current nesting trends, we do not believe the proposed action will cause an appreciable reduction in the likelihood of survival of this species in the wild.

8.3.2 Recovery

The Atlantic recovery plan for the U.S. population of the leatherback sea turtles (NMFS and USFWS 1992) lists the following relevant recovery objective:

> *Objective*: The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico; St. Croix, U.S. Virgin Islands; and along the east coast of Florida.

We believe that the potential loss of up to 4 leatherback sea turtles, and the loss of 6 egg clutches, every 3 years is not likely to have any detectable effect on these nesting trends, we do not believe the proposed action is impeding the progress toward achieving this recovery objective. We also believe that the non-lethal take of 369 leatherback turtles during any consecutive three year period will impact nesting or population trends, as this nonlethal take is not expected to impact these individuals' ability to reproduce. We believe the proposed action is not likely to impede the recovery objective above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. Thus, we believe the proposed action will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild.

8.3.3 Conclusion

The lethal take of 4 and nonlethal take of 369 leatherback sea turtles plus the loss of 6 egg clutches associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of leatherback sea turtles in the wild.

8.4 Loggerhead NWA DPS

The proposed actions covered under this Opinion may result in the following take of loggerhead sea turtles (NWA DPS) over any consecutive 3-year period (Table 46):

- Non-lethal take= 5,270 observed.
- Lethal take= 107 observed and 107 unobserved for a total of 214 lethal takes. The estimated 214 lethal loggerhead sea turtle take include 3.19 sea turtles estimated to be captured during any consecutive three year period that will not be able to be identified when retrieved dismembered.
- 65 lost egg clutches.

8.4.1 Survival

The nonlethal capture of 5,270 NWA DPS loggerhead sea turtles over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of NWA DPS loggerhead sea turtles is anticipated.

The lethal take of 214 loggerhead sea turtles from the NWA DPS over any consecutive 3-year period of the project is a reduction in numbers. As discussed above, lethal interactions would also result in a potential reduction in future reproduction, assuming the individual would be female and would have survived otherwise to reproduce. For this Opinion, we also considered the likelihood of post-interaction mortality to female sea turtles captured during relocation trawling and the likelihood of those females to each loose a single clutch of eggs (65 total for loggerhead sea turtle from the NWA DPS per 3-year period) due to the stress of being captured in a relocation trawling net while gravid (Section 6.1.4.1.2 of this Opinion). While the loss of one clutch would result in the loss of an average of 112.4 eggs, loggerhead sea turtles nest multiple times in a season and therefore would be expected to still lay hundreds of eggs in the same nesting season. The anticipated lethal take of loggerhead sea turtle from the NWA DPS from dredging and relocation trawling, including captures of gravid females in the summer, are expected to occur at discrete project locations within the SARBO action area (i.e., where work will occur) and loggerhead sea turtles generally have large ranges; thus, no reduction in the distribution is expected from the take of these individuals.

Whether or not the reductions in loggerhead sea turtle numbers and reproduction attributed to the proposed action would appreciably reduce the likelihood of survival depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends (i.e., whether the estimated reductions, when viewed within the context of the environmental baseline and status of the species, are of such an extent that adverse effects on population dynamics are appreciable). In Section 4.1.1.5 of this Opinion, we reviewed the status of this species in terms of nesting and female population trends and several recent assessments based on population modeling (i.e., (Conant et al. 2009; NMFS 2009a). Below we synthesize what that information means both in general terms and the more specific context of the proposed action.

Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerhead sea turtles require high survival rates throughout their life to maintain a population. In other words, late-maturing species cannot tolerate much anthropogenic mortality without going into decline. Conant et al. (2009) concluded loggerhead natural growth rates are small, natural survival needs to be high, and even low- to moderate mortality can drive the population into decline. Because recruitment to the adult population is slow, population modeling studies suggest even small increased mortality rates in adults and subadults could substantially impact population numbers and viability (Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1995) (Chaloupka and Musick 1997).

NMFS (2009a) estimated the minimum adult female population size for the NWA DPS in the 2004-2008 timeframe to likely be between approximately 20,000-40,000 individuals (median 30,050), with a low likelihood of being as many as 70,000 individuals. Another estimate for the entire western North Atlantic population was a mean of 38,334 adult females using data from 2001-2010 (Richards et al. 2011). A much less robust estimate for total benthic females in the western North Atlantic was also obtained, with a likely range of approximately 30,000-300,000 individuals, up to less than 1 million.

NMFS (2011c) preliminarily estimated the loggerhead population in the Northwestern Atlantic Ocean along the continental shelf of the Eastern Seaboard during the summer of 2010 at 588,439 individuals (estimate ranged from 382,000 to 817,000) based on positively identified individuals. The NMFS's point estimate increased to approximately 801,000 individuals when including data on unidentified sea turtles that were likely loggerheads. The NMFS (2011c) underestimates the total population of loggerheads since it did not include Florida's east coast south of Cape Canaveral or the Gulf of Mexico, which are areas where large numbers of loggerheads are also expected. In other words, it provides an estimate of a subset of the entire population.

Florida accounts for more than 90% of U.S. loggerhead nesting. The Florida Fish and Wildlife Conservation Commission conducted a detailed analysis of Florida's long-term loggerhead nesting data (1989-2018). The Florida nesting data shown in Figure 28 shows a decline in the number of nests reported annually in the last decade with the lowest reported year occurring in 2007 (28,074 nests), though the number of reported nests has continued to increase since then. Since 1989, all nesting years have reported over nearly 40,000 nests annually except 6 years between 2004-2009 that reported lower numbers of nests. While nest abundance estimates account for only a subset of the entire loggerhead sea turtle population, numbers in the western North Atlantic indicate the population is large (i.e., several hundred thousand individuals). Nesting trends have been significantly increasing over several years against the background of the past and ongoing human and natural factors (as contemplated in the Status of the Species and Environmental Baseline) that have contributed to the current status of the species.

Activities covered under this Opinion could result in the lethal take of 214 loggerhead sea turtles (144 observed, 103 unobserved, and 2.24 observed that cannot be identified to species) and the nonlethal take of 5,270 loggerhead sea turtles and the loss of 65 egg clutches over any consecutive 3-year period of the project; however, we do not expect this loss to result in a detectable change to the population numbers or increasing trends. After analyzing the magnitude of the effects of the proposed action, in combination with the past, present, and future expected impacts to the DPS discussed in this Opinion, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of the loggerhead sea turtle NWA DPS in the wild.

8.4.2 Recovery

The loggerhead recovery plan defines the recovery goal as "…ensur[ing] that each recovery unit meets its Recovery Criteria alleviating threats to the species so that protection under the ESA is no longer necessary" (NMFS and USFWS 2008). The plan then identifies 13 recovery objectives needed to achieve that goal. We do not believe the proposed action impedes the progress of the recovery program or achieving the overall recovery strategy.

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2008) lists the following recovery objectives that are relevant to the effects of the proposed action:

> Objective: Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females.
>
> Objective: Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes.

Recovery is the process of removing threats so self-sustaining populations persist in the wild. The proposed action would not impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy. The recovery plan estimates that the population will reach recovery in 50-150 years following implementation of recovery actions. The minimum end of the range assumes a rapid reversal of the current declining trends; the higher end assumes that additional time will be needed for recovery actions to bring about population growth.

Nesting trends have been increasing since 2007, and take associated with projects covered under this Opinion, especially from hopper dredging, have been and will continue to occur in the action area. While the lethal take of up to 214 loggerhead sea turtles, and the loss of 65 egg clutches over any consecutive 3-year period for projects covered under this Opinion will result in a reduction in numbers and reproduction, it is unlikely to have any detectable influence on the nesting trends noted above. We believe that the non-lethal take of 5,270 loggerhead sea turtles will not impact nesting trends, as non-lethal capture would not impact these individuals' reproduction. Given nesting populations are steadily increasing, the projected take is not expected to have any discernable impact to the species.

8.4.3 Conclusion

The lethal take of 214 and nonlethal take of 5,270 loggerhead sea turtles plus the loss of 65 egg clutches associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of loggerhead sea turtles in the wild.

8.5 Atlantic Sturgeon (All 5 DPSs)

The proposed actions covered under this Opinion may result in the following take of Atlantic sturgeon over any consecutive 3-year period (Table 51):

- Nonlethal take= 942 observed.
- Lethal take= 71 observed and 66 unobserved for a total of 137 lethal take.

Because subadult and adult Atlantic sturgeon mix extensively in the marine and estuarine environments, individuals from all 5 Atlantic sturgeon DPSs could occur within the action area.

Therefore, we must determine from which DPSs the takes will occur. As described in Section 4.1.2.3, USGS has recently completed a draft mixed stock analysis specific to the Southeast Region (USGS unpublished data). We used the information from that report to calculate an estimate of the likely number of Atlantic sturgeon occurring the Southeast (North Carolina/Virginia Border to Florida). Table 31 reports the likely DPS composition for individuals in the Southeast as reported in the mixed stock analysis (USGS unpublished data) and our estimates of the likely minimum number of individuals from each DPS occurring in the Southeast.

DPS Composition and Minimum Number of Individuals in the Southeast from Table 31

Proportion of Individuals from Each DPS	Minimum Number of Individuals in Southeast by DPS
South Atlantic – 52.9% (49.9%-57.0%)	5,067
Carolina – 33.8% (29.2%-36.4%)	3,243
Chesapeake – 9.6% (7.9%-12.1%)	920
New York Bight – 3.6% (2.5%-4.8%)	343
Gulf of Maine – 1.0% (0-0.4%)	9

Table 51 shows the breakdown of anticipated lethal and nonlethal take by DPS over any consecutive 3-year period of the project based on the DPS composition in the Southeast. Because you cannot incidentally take just a portion of an animals, these estimate are rounded up to the nearest whole number. This approach is conservative toward the species; however, in instances where low levels are take are anticipated, it can significantly overestimate the potential impacts of an action. For example, we estimated 0.14 animals from Gulf of Maine DPS maybe killed every 3 years or 0.05 annually. This equates to the mortality of 1 individual every 20 years. By rounding up to the nearest whole number, that estimate changes to 1 mortality every 3 years.

Table 51. Anticipated Take by DPS Over any Consecutive 3-year Period of the Project

Atlantic Sturgeon DPS	Total Anticipated Take (All DPSs Combined)	% Composition in Southeast	Take by DPS	Take by DPS (rounded up)
Lethal				
South Atlantic (SA)	137	52.9%	72.47	73
Carolina	137	33.8%	46.31	47
Chesapeake Bay	137	9.6%	13.15	14
New York Bight	137	3.6%	4.93	5
Gulf of Maine	137	0.1%	0.14	1
Nonlethal				
South Atlantic	942	52.9%	498.32	499
Carolina	942	33.8%	318.40	319
Chesapeake Bay	942	9.6%	90.43	91
New York Bight	942	3.6%	33.91	34
Gulf of Maine	942	0.1%	0.94	1

8.5.1 SA DPS

8.5.1.1 SA DPS Survival

The proposed action may result in 572 Atlantic sturgeon takes from the SA DPS over consecutive 3-year periods. We estimate those takes would be 73 lethal and 499 nonlethal (Table 51). Based on most recent draft MSA conducted for the Southeast, we expect that 52.9% of the Atlantic sturgeon in the action area will originate from the SA DPS (USGS unpublished data). The nonlethal capture of 499 individuals from the SA DPS over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential lethal take of 73 SA DPS Atlantic sturgeon over consecutive 3-year periods would reduce the population of Atlantic sturgeon in the SA DPS by that amount. Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in Georgia prior to 1890. At the time of listing, only 6 spawning subpopulations were believed to have existed in the SA DPS: Combahee River, Edisto River, Savannah River, Ogeechee River, Altamaha River (including the Oconee and Ocmulgee tributaries), and Satilla River. Three of the spawning subpopulations in the SA DPS are relatively robust and are considered the second (Altamaha River) and third (Combahee/Edisto River) largest spawning subpopulations across all 5 DPSs. Peterson et al. (2008) estimated the number of spawning adults in the Altamaha River was 324 (95% CI: 143-667) in 2004 and 386 (95% CI: 216-787) in 2005. Bahr and Peterson (2016) estimated the age-1 juvenile abundance in the Savannah River from 2013-2015 at 528 in 2013, 589 in 2014, and 597 in 2015. As described in the Status of the Species, and based largely on these studies, we estimate 5,067 individuals (Table 31) from the SA DPS are likely to be in the Southeast. NMFS Greater Atlantic Region estimated the ocean population of the SA DPS is 14,911 (Table 30).

We anticipate 73 mortalities from the SA DPS during consecutive 3-year periods (Table 51) is unlikely to change its status, as this loss represents a small percentage of the SA DPS population as a whole. Based on our estimates of 5,067 individuals from SA DPS in the Southeast (Table 31), the mortality of 73 individuals from the SA DPS every 3 years would represent 1.44% of the SA DPS population occurring in the Southeast. Compared to the SA DPS ocean population of 14,911 (Table 30), estimated by NMFS Greater Atlantic Region, the loss of 71 individuals from the, would only represents 0.49% of the population. The best available information on the status of the SA DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (ASMFC 2017). The assessment determined the SA DPS abundance is "depleted" relative to historical levels. The assessment concluded there was not enough information available to assess the abundance of the DPS since the implementation of the 1998 fishing moratorium.

Both our estimates of the Atlantic sturgeon SA DPS in the Southeast and the NMFS Greater Atlantic Region's estimates represent only a percentage of the total SA DPS population as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. However, it is also worth noting the activities included in the proposed action have been on-going for many years in the action area, so we believe these lethal takes are unlikely to represent new sources of mortality for animals of the SA DPS. Instead, these takes are likely a more accurate reflection of the takes that have been on-going over the last several years. Because our take estimates use data collected in years prior to the implementation of the Sturgeon PDCs, we anticipate that the PDCs will reduce the impact of take from the proposed action relative to historical levels. The actual number of mortalities occurring in the future may be less than what we have estimated here.

The loss of 73 individuals over any consecutive 3-year period may affect the reproductive potential of the SA DPS. We anticipate these takes could be of individuals from any sex or age class of the SA DPS population. Some of the dredging covered under this Opinion will occur in rivers supporting all life stages of Atlantic sturgeon, though the PDCs will prohibit dredging in areas where eggs and larvae may occur. Therefore, juveniles, subadults, and adults could make up the mortalities. The potential loss of sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles/subadults would be less, because they are generally more abundant than adults and are not yet sexually mature. Additionally, because of the requirements outlined in the Sturgeon PDCs the proposed action will also not affect the spawning grounds within SA DPS.

The mortalities associated with the proposed action is not likely to reduce distribution of the SA DPS. We anticipate a up to 1 individual could be removed from the Savannah River over any consecutive 3-year period during cutterhead dredging, but we do not anticipate this mortality would have any measurable effect on the overall distribution of the animals from the DPS. We anticipate the other mortalities occurring over any consecutive 3-year period could take place anywhere in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the nonlethal takes expected and the death of up to 73 individuals from the SA DPS Atlantic sturgeon during any consecutive 3-year period of the project, will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect SA DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the death of 73 SA

DPS Atlantic sturgeon represents a small percentage of the total population of the DPS; those deaths are unlikely to change the status or trends of the DPS as a whole; the loss of those individuals are likely to have a small effect on reproductive output; and the action will have only a minor and temporary effect on the distribution of SA DPS Atlantic sturgeon in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the SA DPS will survive in the wild.

8.5.1.2 SA DPS Recovery

A Recovery Plan for the SA DPS has not yet been developed. However, NMFS completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017f). The final listing rule (77 FR 5914; Publication Date February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as on-going threats. The severity of those threats varies by DPS.

While the proposed action includes dredging, which is considered one of the major threats to Atlantic sturgeon, we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the SA DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the SA DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

Habitat will be disturbed by the maintenance dredging considered here, but we do not believe these disturbances will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. The Sturgeon PDCs were specifically developed to protect spawning habitats and ensure habitat connectivity by requiring that at least 50% of migratory habitat are always available and free of dredge-related obstructions. Similarly, the Sturgeon PDCs were specifically designed to protect sturgeon when environmental conditions are degraded and naturally stressful for all life stages. Once the maintenance dredging considered in this Opinion is completed for a given river, we do not anticipate that any impacts to habitat caused by the proposed action will affect how sturgeon use the action area.

With these conservative measures in place, we anticipate mortality rates will be greatly reduced from what might otherwise be expected. Nonetheless, the proposed action will result in a mortalities during consecutive 3-year periods, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers will be small relative to remaining population and as a result the impact on reproduction and future year classes will also be small enough not to affect the status of the DPS. Accordingly, we do not believe the proposed action will impede the recovery of the SA DPS, by significantly exacerbating the dredging effects of any of the other remaining major threats identified in the final listing rules. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the SA DPS.

8.5.2 Carolina DPS

8.5.2.1 Carolina DPS Survival

The proposed action may result in 366 Atlantic sturgeon takes from the Carolina DPS over consecutive 3-year periods. We estimate those takes would be 47 lethal and 319 nonlethal (Table 51). Based on most recent draft MSA conducted for the Southeast, we expect that 33.8% of the Atlantic sturgeon in the action area will originate from the Carolina DPS (USGS unpublished data). The nonlethal capture of 319 individuals from the Carolina DPS over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential lethal take of 47 Atlantic sturgeon over consecutive 3-year periods would reduce the population of Atlantic sturgeon in the Carolina DPS by that amount. Historical fishery landings data indicate between 7,000 and 10,500 adult female Atlantic sturgeon were present in North Carolina prior to 1890 (Armstrong and Hightower 2002; Secor 2002). Secor (2002) estimates that 8,000 adult females were present in South Carolina during that same time-frame. At the time of listing, the abundance for each river population within the DPS was estimated to have fewer than 300 spawning adults; estimated to be less than 3% of what they were historically (Atlantic Sturgeon Status Review Team 2007).

We estimate 47 mortalities from the Carolina DPS during consecutive 3-year periods (Table 51) is unlikely to change its status, as this loss represents a small percentage of the Carolina DPS population as a whole. Based on our estimates of 3,243 individuals from Carolina DPS in the Southeast (Table 31), the mortality of 47 individuals from the Carolina DPS every 3 years would represent 1.5% of the Carolina DPS population occurring in the Southeast. Compared to the Carolina DPS ocean population of 1,356 (Table 30), estimated by NMFS Greater Atlantic Region, the loss of 47 individuals from the, would represent 3.5% of the population. However, as discussed in Section 4.1.2.3, we believe that the Greater Atlantic Region's estimate does not accurately reflect the likely population abundance of individuals from Carolina DPS, given the distance from the Carolina DPS where those samples were collected of the sampling used. The best available information on the overall trend of the Carolina DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (Atlantic States Marine Fisheries Commission 2017), discussed in Section 4.1.2.3. The Assessment determined the Carolina DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (67%) that the Carolina DPS abundance has increased since the implementation of the 1998 fishing moratorium.

Both our estimates of the Atlantic sturgeon Carolina DPS in the Southeast and the NMFS Greater Atlantic Region's estimates represent only a percentage of the total Carolina DPS population as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. However, it is also worth noting the activities included in the proposed action have been on-going for many years in the action area, so we believe these lethal takes are unlikely to represent new sources of mortality for animals of the Carolina DPS. Instead, these takes are likely a more accurate reflection of the takes that have been on-going over the last several years. Additionally, because our take estimates use data collected in years prior to the implementation of the Sturgeon PDCs, we anticipate that the PDCs may reduce the impact of take from the proposed action relative to historical levels.

We do not believe the loss of 47 individuals every 3 years will meaningful affect the reproductive potential of the Carolina DPS. We anticipate these takes could be of individuals from any sex or age class of the Carolina DPS population. Thus, we do not anticipate any portion of the Carolina DPS would be disproportionately affected by the action. Some of the dredging covered under this Opinion will occur in rivers supporting all life stages of Atlantic sturgeon, though the PDCs will prohibit dredging in areas where eggs and larvae may occur. Therefore, juveniles, subadults, and adults could make up the mortalities. The potential loss of sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles/subadults would generally be less, all things being equal, because they are generally more abundant than adults and are not yet sexually mature. Additionally, because of the requirements outlined in the Sturgeon PDCs the proposed action will also not affect the spawning grounds within Carolina DPS.

The mortalities associated with the proposed action is not likely to reduce distribution of the Carolina DPS. We anticipate cutterhead dredging could remove up to 1 individual from the Cooper River and 1 individual from the Cape Fear River during any consecutive 3-year period. We do not anticipate these potential removals would have any measurable effect on the overall distribution of the animals from the DPS. We anticipate the other mortalities occurring over any consecutive 3-year period could take place anywhere in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the nonlethal takes expected and the death of up to 47 individuals from the Carolina DPS Atlantic sturgeon during any consecutive 3-year period of the project, will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect Carolina DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the death of 47 Carolina DPS Atlantic sturgeon represents a small percentage of the total population of the DPS; those deaths are unlikely to change the status or trends of the DPS as a whole; the loss of those individuals are likely to have a small effect on reproductive output; and the action will have only a minor and temporary effect on the distribution of Carolina DPS Atlantic sturgeon in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated lethal takes will appreciably reduce the likelihood that the Carolina DPS will survive in the wild.

8.5.2.2 Carolina DPS Recovery

A Recovery Plan for the Carolina DPS has not yet been developed. However, NMFS completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017f). The final listing rule (77 FR 5914; Publication Date February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as on-going threats. The severity of those threats varies by DPS.

While the proposed action includes dredging, which is considered one of the major threats to Atlantic sturgeon, we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the Carolina DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the Carolina DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

Habitat will be disturbed by the maintenance dredging considered here, but we do not believe these disturbances will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. The Sturgeon PDCs were specifically developed to protect spawning habitats and ensure habitat connectivity by requiring that at least 50% of migratory habitat are always available and free of dredge-related obstructions. Similarly, the Sturgeon PDCs were specifically designed to protect sturgeon when environmental conditions are degraded and naturally stressful for all life stages. Once the maintenance dredging considered in this Opinion is completed for a given river, we do not anticipate that any impacts to habitat caused by the proposed action will affect how sturgeon use the action area.

With these conservative measures in place, we anticipate mortality rates will be reduced from what might otherwise be expected. Nonetheless, the proposed action will result in a mortalities during consecutive 3-year periods, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers will be small relative to remaining population and as a result the impact on reproduction and future year classes will also be small enough not to affect the trend of the DPS. Accordingly, we do not believe the proposed action will impede the recovery of the Carolina DPS, by significantly exacerbating the dredging effects of any of the other remaining major threats identified in the final listing rules. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the Carolina DPS.

8.5.3 Chesapeake Bay DPS

8.5.3.1 Chesapeake Bay DPS Survival

The proposed action may result in 105 Atlantic sturgeon takes from the Chesapeake Bay DPS over consecutive 3-year periods. We estimate those takes would be 14 lethal and 91 nonlethal (Table 51). Based on most recent draft MSA conducted for the Southeast, we expect that 9.6% of the Atlantic sturgeon in the action area will originate from the Chesapeake Bay DPS (USGS unpublished data). The nonlethal capture of 91 individuals from the Chesapeake Bay DPS over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential lethal take of 14 Atlantic sturgeon over consecutive 3-year periods would reduce the population of Atlantic sturgeon in the Chesapeake Bay DPS by that amount. Historically, the Chesapeake Bay DPS likely supported more than 10,000 spawning adults (Atlantic Sturgeon Status Review Team 2007; Maine State Planning Office 1993; Secor 2002). Currently, there are 4 known spawning subpopulations for the Chesapeake Bay DPS, one each for the Pamunkey River and for Marshyhope Creek, and 2 for the James River (Balazik et al. 2017; Balazik et al. 2012a) (Balazik and Musick 2015; Greenlee and Secor 2016; Hager et al. 2014; Richardson and Secor 2017).

We estimate 14 mortalities from the Chesapeake Bay DPS during consecutive 3-year periods (Table 51) is unlikely to change its status, as this loss represents a small percentage of the Chesapeake Bay DPS population as a whole. As noted in Section 4.1.2.3, our current understanding of the migratory behavior of Atlantic sturgeon suggests these animals would be transient individuals that are unlikely to represent a significant portion of the total population from the DPS. Based on our estimates of 920 individuals from Chesapeake Bay DPS in the Southeast (Table 31), the mortality of 14 individuals from the Chesapeake Bay DPS every 3 years would represent 1.5% of the Chesapeake Bay DPS population estimated to occur in the Southeast. Compared to the Chesapeake Bay DPS ocean population of 8,811 (Table 30), estimated by NMFS Greater Atlantic Region, the loss of 14 individuals, would only represents 0.2% of the population. Based on the discussion in Section 4.1.2.3, we believe the NMFS Greater Atlantic Region estimate is based on information that more accurately reflects the likely population abundance of individuals from this DPS. The best available information on the overall trend of the Chesapeake Bay DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (Atlantic States Marine Fisheries Commission 2017), discussed in Section 4.1.2.3. The Assessment determined the Chesapeake Bay DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively low probability (37%) that the Chesapeake Bay DPS abundance has increased since the implementation of the 1998 fishing moratorium (Atlantic States Marine Fisheries Commission 2017).

Both our estimates of the Atlantic sturgeon Chesapeake Bay DPS in the Southeast and the NMFS Greater Atlantic Region's estimates represent only a percentage of the total Chesapeake Bay DPS population as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. However, it is also worth noting the activities included in the proposed action have been on-going for many years in the action area, so we believe these lethal takes are unlikely to represent new sources of mortality for animals of the Chesapeake Bay DPS. Instead, these takes are likely a more accurate reflection of the takes that have been on-going over the last several years. Because our take estimates use data collected in years prior to the implementation of the Sturgeon PDCs, we anticipate the actual number of mortalities occurring in the future may be less than what we have estimated here.

We do not believe the loss of 14 individuals every 3 years will meaningful affect the reproductive potential of the Chesapeake Bay DPS. Given the distance between spawning rivers of the Chesapeake Bay DPS to the action area, we anticipate subadults or adults are the most likely life stages to be lethally taken. We anticipate these takes could be of individuals from any sex. The potential loss of sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. We anticipate the overall impact to the population as whole from the loss of subadults would generally be less, because they are generally more abundant than adults and are not yet sexually mature. Additionally, because of the requirements outlined in the Sturgeon PDCs the proposed action will also not affect the spawning grounds within Chesapeake Bay DPS.

The mortalities associated with the proposed action is not likely to reduce distribution of the Chesapeake Bay DPS. The removal of these individuals could take place anywhere in the action area. Thus, we do anticipate any changes of the distribution of the species range wide. Similarly, we do not believe any particular riverine population within the DPS will be disproportionally affected by mortalities, particularly due to the expected transient nature of Chesapeake Bay DPS individuals in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the nonlethal takes expected and the death of up to 14 individuals from the Chesapeake Bay DPS Atlantic sturgeon during any consecutive 3-year period of the project, will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect Chesapeake Bay DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the death of 14 Chesapeake Bay DPS Atlantic sturgeon represents a small percentage of the total population of the DPS; those deaths are unlikely to change the status or trends of the DPS as a whole; the loss of those individuals are likely to have a small effect on reproductive output; and the action will have only a minor and temporary effect on the distribution of Chesapeake Bay DPS Atlantic sturgeon in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the Chesapeake Bay DPS will survive in the wild.

8.5.3.2 Chesapeake Bay DPS Recovery

A Recovery Plan for the Chesapeake Bay DPS has not yet been developed. However, NMFS completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017f). The final listing rule (77 FR 5914; Publication Date February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as on-going threats. The severity of those threats varies by DPS.

While the proposed action includes dredging, which is considered one of the major threats to Atlantic sturgeon, we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the Chesapeake Bay DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the Chesapeake Bay DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

Habitat will be disturbed by the maintenance dredging considered here, but we do not believe these disturbances will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. The Sturgeon PDCs were specifically developed to ensure habitat connectivity by requiring that at least 50% of migratory habitat are always available and free of dredge-related obstructions. Similarly, the Sturgeon PDCs were specifically designed to protect sturgeon when environmental conditions are degraded and naturally stressful for all life stages. Once the maintenance dredging considered in this Opinion is completed for a given river, we do not anticipate that any impacts to habitat caused by the proposed action will affect how sturgeon use the action area. Because individuals from this DPS are unlikely to spawn in the action area, the proposed action is unlikely to have any effect on their spawning habitat.

With these conservative measures in place, we anticipate mortality rates will be reduced from what might otherwise be expected. Nonetheless, the proposed action will result in a mortalities during consecutive 3-year periods, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers will be small relative to remaining population and as a result the impact on reproduction and future year classes will also be small enough not to affect the status of the DPS. Accordingly, we do not believe the proposed action will impede the recovery of the Chesapeake Bay DPS, by significantly exacerbating the dredging effects of any of the other remaining major threats identified in the final listing rules. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the Chesapeake Bay DPS.

8.5.4 New York Bight DPS

8.5.4.1 New York Bight DPS Survival

The proposed action may result in 39 Atlantic sturgeon takes from the New York Bight DPS over consecutive 3-year periods. We estimate those takes would be 5 lethal and 34 nonlethal (Table 51). Based on most recent draft MSA conducted for the Southeast, we expect that 3.6% of the Atlantic sturgeon in the action area will originate from the New York Bight DPS (USGS unpublished data). The nonlethal capture of 34 individuals from the New York Bight DPS over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential lethal take of 5 Atlantic sturgeon over consecutive 3-year periods would reduce the population of Atlantic sturgeon in the New York Bight by that amount. Prior to the onset of expanded fisheries exploitation of sturgeon in the 1800s, a conservative historical estimate for the Hudson River Atlantic sturgeon population was 10,000 adult females (Secor 2002). Current population abundance is likely at least one order of magnitude smaller than historical levels (Atlantic Sturgeon Status Review Team 2007; Kahnle et al. 2007; Secor 2002). Based on data collected from 1985-1995, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population (Kahnle et al. 2007). There is no abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800s indicate that this was historically a large population, with an estimated 180,000 adult females prior to 1890 (Secor 2002; Secor and Waldman 1999). Based on the capture of juvenile Atlantic sturgeon in the Delaware River, researchers estimated estimate there were 3,656 (95% CI = 1,935–33,041) age 0-1 juvenile Atlantic sturgeon in the Delaware River subpopulation in 2014 (Hale et al. 2016). However, the relatively low numbers of captured adults suggest the existing riverine subpopulation is limited in size. For example, of the 261 adult-sized Atlantic sturgeon captured for scientific purposes off the Delaware Coast between 2009 and 2012, 100 were subsequently identified by genetics analysis to belong to the Hudson River subpopulation while only 36 belonged to the Delaware River subpopulation (Wirgin et al. 2015). The Atlantic Sturgeon Status Review Team (2007) suggested that there may be less than 300 spawning adults per year for the Delaware River portion of the New York Bight DPS.

We estimate 5 mortalities from the New York Bight DPS during consecutive 3-year periods (Table 51) is unlikely to change its status, as this loss represents a small percentage of the e New York Bight DPS population as a whole. As noted in Section 4.1.2.3, our current understanding of the migratory behavior of Atlantic sturgeon suggests these animals would be transient individuals that are unlikely to represent a significant portion of the total population from the DPS. Based on our estimates of 343 individuals from New York Bight DPS in the Southeast (Table 31), the mortality of 5 individuals from the New York Bight DPS every 3 years would represent 1.45% of the New York Bight DPS population occurring in the Southeast. Compared to the New York Bight DPS ocean population of 34,566 (Table 30), estimated by NMFS Greater Atlantic Region, the loss of 5 individuals, would only represents 0.01% of the population. Based on the discussion in Section 4.1.2.3, we believe the NMFS Greater Atlantic Region estimate is based on information that more accurately reflects the likely population abundance of individuals from this DPS. The best available information on the overall trend of the New York Bight DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (Atlantic States Marine Fisheries Commission 2017), discussed in Section 4.1.2.3. The 2017 Assessment determined the New York Bight DPS abundance is "depleted" relative to historical levels. It also determined there is a relatively high probability (75%) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium (Atlantic States Marine Fisheries Commission 2017).

Both our estimates of the Atlantic sturgeon New York Bight DPS in the Southeast and the NMFS Greater Atlantic Region’s estimates represent only a percentage of the total New York Bight DPS population as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. However, it is also worth noting the activities included in the proposed action have been on-going for many years in the action area, so we believe these lethal takes are unlikely to represent new sources of mortality for animals of the New York Bight DPS. Instead, these takes are likely a more accurate reflection of the takes that have been on-going over the last several years. Because our take estimates use data collected in years prior to the implementation of the Sturgeon PDCs, we anticipate the actual number of mortalities occurring in the future may be less than what we have estimated here.

We do not believe the loss of 6 individuals every 3 years will meaningful affect the reproductive potential of the New York Bight DPS. Given the distance of the spawning rivers of the New York Bight DPS to the action area, we anticipate subadults or adults are the most likely life stages to be lethally taken. We anticipate these takes could be of individuals from any sex. The potential loss of sexually mature female would preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. We anticipate the overall impact to the population as whole from the loss of subadults would generally be less, all things being equal, because they are generally more abundant than adults and are not yet sexually mature. Additionally, because of the requirements outlined in the Sturgeon PDCs the proposed action will also not affect the spawning grounds within New York Bight DPS.

The mortalities associated with the proposed action is not likely to reduce distribution of the New York Bight DPS. The removal of these individuals could take place anywhere in the action area. Thus, we do anticipate any changes of the distribution of the species range wide. Similarly, we do not believe any particular riverine population within the DPS will be disproportionally affected by mortalities, particularly due to the expected transient nature of the New York Bight DPS individuals in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the nonlethal takes expected and the death of up to 6 individuals from the New York Bight DPS Atlantic sturgeon during any consecutive 3-year period of the project, will not appreciably reduce the likelihood of survival of the DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect New York Bight DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the death of 6 New York Bight DPS Atlantic sturgeon represents a small percentage of the total population of the DPS; those deaths are unlikely to change the status or trends of the DPS as a whole; the loss of those individuals are likely to have a small effect on reproductive output; and the action will have only a minor and temporary effect on the distribution of New York Bight DPS Atlantic sturgeon in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the New York Bight DPS will survive in the wild.

8.5.4.2 New York Bight DPS Recovery

A Recovery Plan for the New York Bight DPS has not yet been developed. However, NMFS completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017f). The final listing rule (77 FR 5914; Publication Date February 6, 2012) identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as on-going threats. The severity of those threats varies by DPS.

While the proposed action includes dredging, which is considered one of the major threats to Atlantic sturgeon, we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the New York Bight DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the New York Bight DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

Habitat will be disturbed by the maintenance dredging considered here, but we do not believe these disturbances will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. The Sturgeon PDCs were specifically developed to ensure habitat connectivity by requiring that at least 50% of migratory habitat are always available and free of dredge-related obstructions. Similarly, the Sturgeon PDCs were specifically designed to protect sturgeon when environmental conditions are degraded and naturally stressful for all life stages. Once the maintenance dredging considered in this Opinion is completed for a given river, we do not anticipate that any impacts to habitat caused by the proposed action will affect how sturgeon use the action area. Because individuals from this DPS are unlikely to spawn in the action area, the proposed action is unlikely to have any effect on their spawning habitat.

With these conservative measures in place, we anticipate mortality rates will be greatly reduced from what might otherwise be expected. Nonetheless, the proposed action will result in a mortalities during consecutive 3-year periods, leading to a subsequent relatively small reduction in future reproductive output. This reduction in numbers will be small relative to remaining population and as a result the impact on reproduction and future year classes will also be small enough not to affect the trend of the DPS. Accordingly, we do not believe the proposed action will impede the recovery of the New York Bight DPS, by significantly exacerbating the dredging effects of any of the other remaining major threats identified in the final listing rules. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the New York Bight DPS.

8.5.5 Gulf of Maine DPS

8.5.5.1 Gulf of Maine DPS Survival

The proposed action may result in 2 Atlantic sturgeon takes from the Gulf of Maine DPS over consecutive 3-year periods. We estimate those takes would be 1 lethal and 1 nonlethal. Based on most recent draft MSA conducted for the Southeast, we expect that 0.1% of the Atlantic sturgeon in the action area will originate from the Gulf of Maine DPS (USGS unpublished data). The nonlethal capture of 1 individual (Table 51) from the Gulf of Maine DPS over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this DPS. We anticipate this individual will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of Atlantic sturgeon is anticipated.

The potential lethal take of 1 Atlantic sturgeon (adult) over consecutive 3-year periods would reduce the population of Atlantic sturgeon in the Gulf of Maine DPS by that amount. Historically, the Gulf of Maine DPS likely supported more than 10,000 spawning adults (Atlantic Sturgeon Status Review Team 2007; Maine State Planning Office 1993; Secor 2002). The Atlantic Sturgeon Status Review Team (2007) presumed that the Gulf of Maine DPS was comprised of less than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over 2 time periods, 1977-1981 and 1998-2000, resulted in the capture of 9 adult Atlantic sturgeon (Squiers 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the gear used may not have been selective for larger, adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies.

We estimate 1 mortality from the Gulf of Maine DPS will occur during consecutive 3-year periods (Table 51). The death of this individual over this period will reduce the total number animals in the Gulf of Maine DPS compared to the number that would have been present without the proposed action. Based on our estimates of 9 individuals from Gulf of Maine DPS in the Southeast (Table 31), the mortality of 1 individual from the Gulf of Maine DPS every 3 years would represent 11.1% of the Gulf of Maine DPS population occurring in the Southeast. However, it is unlikely this reduction in numbers will change the status of this DPS as this loss represents a small percentage of the Gulf of Maine DPS population as a whole. As noted in Section 4.1.2.3, our current understanding of the migratory behavior of Atlantic sturgeon suggests this animal would be a transient individual that is unlikely to represent a significant portion of the total population from the DPS. Compared to the Gulf of Maine DPS ocean population of 7,455 (Table 30), estimated by NMFS Greater Atlantic Region, the loss of 1 individual, would only represents 0.01% of the population. Based on the discussion in Section 4.1.2.3, we believe the NMFS Greater Atlantic Region estimate is based on information that more accurately reflects the likely population abundance of individuals from this DPS. The best available information on the overall trend of the Gulf of Maine DPS comes from the 2017 Atlantic Sturgeon Benchmark Stock Assessment (Atlantic States Marine Fisheries Commission 2017), discussed in Section 4.1.2.3. The 2017 Assessment determined the Gulf of Maine DPS abundance is "depleted" relative to historical levels. It also determined there is a 51% probability Gulf of Maine DPS abundance has increased since the implementation of the 1998 fishing moratorium (Atlantic States Marine Fisheries Commission 2017).

Both our estimates of the Atlantic sturgeon Gulf of Maine DPS in the Southeast and the NMFS Greater Atlantic Region’s estimates represent only a percentage of the total Gulf of Maine DPS population as they do not include all individuals from all age classes, meaning the absolute population abundance is higher. While some information is available on how individual riverine populations within the DPS are faring over time, we do not have information regarding the overall population trends of the DPS as a whole. However, it is also worth noting the activities included in the proposed action have been on-going for many years in the action area, so we believe these lethal takes are unlikely to represent new sources of mortality for animals of the

Gulf of Maine DPS. Instead, these takes are likely a more accurate reflection of the takes that have been on-going over the last several years. Because our take estimates use data collected in years prior to the implementation of the Sturgeon PDCs, we anticipate the actual number of mortalities occurring in the future may be less than what we have estimated here.

We do not believe the loss of 1 individual every 3 years will meaningful affect the reproductive potential of the Gulf of Maine DPS. Given the distance of the spawning rivers of the Gulf of Maine DPS to the action area, we anticipate subadults or adults are the most likely life stages to be lethally taken. We anticipate these takes could be of individuals from any sex. If that loss was a female, the mortality could preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Additionally, the proposed action will also not affect the spawning grounds within Gulf of Maine DPS.

The mortalities associated with the proposed action is not likely to reduce distribution of the Gulf of Maine DPS. The removal of these individuals could take place anywhere in the action area. Thus, we do anticipate any changes of the distribution of the species range wide. Similarly, we do not believe any particular riverine population within the DPS will be disproportionally affected by mortalities, particularly due to the expected transient nature of Gulf of Maine DPS individuals in the action area. Therefore, we do not believe the overall distribution of the DPS will be affected by the proposed action.

Based on the information provided above, the nonlethal takes expected and the death of no more than 1 individual from the Gulf of Maine DPS Atlantic sturgeon during any consecutive 3-year period of the project, will not appreciably reduce the likelihood of survival of the Gulf of Maine DPS (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect Gulf of Maine DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the death of 1 Gulf of Maine DPS Atlantic sturgeon represents an extremely small percentage of the population of the DPS; that death is unlikely to change the status or trends of the DPS as a whole; the loss of that individual is likely to have a small effect on reproductive output; and the action will have only a minor and temporary effect on the distribution of Gulf of Maine DPS Atlantic sturgeon in the action area and no effect on the distribution of the DPS throughout its range. Therefore, we do not believe the anticipated takes will appreciably reduce the likelihood that the Gulf of Maine DPS will survive in the wild.

8.5.5.2 Gulf of Maine DPS Recovery

A Recovery Plan for the Gulf of Maine DPS has not yet been developed. However, NMFS completed a recovery outline for Atlantic sturgeon in 2017 (NMFS 2017f). The final list rule identified threats to all 5 DPSs as including: dams, dredging, water quality, climate change, and overutilization for commercial purposes. The recovery outline indicates those threats are still largely of concern and further identifies habitat changes; impeded access to historical habitat by dams and reservoirs; degraded water quality; reduced water quantity; vessel strikes; and bycatch in commercial fisheries as on-going threats. The severity of those threats varies by DPS.

While the proposed action includes dredging, which is considered one of the major threats to Atlantic sturgeon, we do not anticipate the effects from the proposed action will impede recovery. In general, to recover, a listed species must have sustained population growth. For the Gulf of Maine DPS to exhibit sustained population growth, there must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. Environmental conditions must be suitable for the successful development and growth of all life stages, particularly the most vulnerable early life stages. Mortality rates at all life stages must be low enough to ensure successful recruitment of individuals into subsequent age classes so that successful spawning can continue over time and over generations. For the Gulf of Maine DPS, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness.

Habitat will be disturbed by the maintenance dredging considered here, but we do not believe these disturbances will significantly or permanently reduce suitable habitat for spawning, foraging, resting and migrations of all individuals. The Sturgeon PDCs were specifically developed to ensure habitat connectivity by requiring that at least 50% of migratory habitat are always available and free of dredge-related obstructions. Similarly, the Sturgeon PDCs were specifically designed to protect sturgeon when environmental conditions are degraded and naturally stressful for all life stages. Once the maintenance dredging considered in this Opinion is completed for a given river, we do not anticipate that any impacts to habitat caused by the proposed action will affect how sturgeon use the action area. Because individuals from this DPS are unlikely to spawn in the action area, the proposed action is unlikely to have any effect on their spawning habitat.

With these conservative measures in place, we anticipate mortality rates will be greatly reduced from what might otherwise be expected. Nonetheless, the proposed action will result in a small amount of mortality during consecutive 3-year periods (1 lethal take), leading to a subsequent small reduction in future reproductive output. This reduction in numbers will be small relative to remaining population and as a result the impact on reproduction and future year classes will also be small enough not to affect the trend of the DPS. Accordingly, we do not believe the proposed action will impede the recovery of the Gulf of Maine DPS, by significantly exacerbating the dredging effects of any of the other remaining major threats identified in the final listing rules. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery of the Gulf of Maine DPS.

8.5.6 Conclusion

The lethal take of 137 and nonlethal take of 942 Atlantic sturgeon associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of any of the 5 Atlantic sturgeon DPSs in the wild.

8.6 Shortnose Sturgeon

The proposed actions covered under this Opinion may result in the following take of shortnose sturgeon over any consecutive 3-year period (Table 46):

- Non-lethal take= 6 observed.
- Lethal take= 8 observed and 6 unobserved for a total of 14 lethal take.

8.6.1 Survival

The proposed action may result in 20 shortnose sturgeon takes over consecutive 3-year periods. We estimate those takes would be 14 lethal and 6 nonlethal. The nonlethal capture of 6 individuals (Table 46) over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of the species. We anticipate these individuals will fully recover such that no reductions in reproduction or numbers are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of shortnose sturgeon is anticipated.

The potential lethal take of 14 shortnose sturgeon over consecutive 3-year periods would reduce the population of shortnose sturgeon by that amount. There are no reliable estimates of the size of the shortnose sturgeon population as a whole. As discussed in Section 4.1.3 of this Opinion, shortnose sturgeon in the Southeast comprise a single metapopulation, the "Carolinian Province" (Figure 30) and that individuals within this metapopulation show less reproductive isolation between rivers than the other 2 shortnose sturgeon metapopulations. The status of the shortnose sturgeon in the Southeast is mixed. Populations within the southern metapopulation are relatively small compared to their northern counterparts. The Altamaha River supports the largest known shortnose sturgeon population in the Southeast with successful self-sustaining recruitment. Total population estimates in the Altamaha show large interannual variation is occurring; estimates have ranged from as low as 468 fish in 1993 to over 5,550 fish in 2006 (NMFS 1998; Peterson and Bednarski 2013). Abundance estimates for the Ogeechee River indicate the shortnose sturgeon population in this river is considerably smaller than in the Altamaha River. The highest point estimate since 1993 occurred in 2007 and resulted in a total Ogeechee River population estimate of 404 shortnose sturgeon (95% confidence interval [CI]: 175-633) (Peterson and Farrae 2011). However, subsequent sampling in 2008 and 2009 resulted in point estimates of 264 (95% CI: 126-402) and 203 (95% CI: 32-446), respectively (Peterson and Farrae 2011). Spawning is also occurring in the Savannah, Cooper, Congaree, and Yadkin-Pee Dee Rivers. The Savannah River shortnose sturgeon population is possibly the second largest in the Southeast with highest point estimate of the total population occurring in 2013 at 2,432 (95% CI: 1,025-6,102). Mean population estimates were lower in 2014 and 2015, reaching an estimated 1,390 (95% CI: 890-2,257) total individuals in 2015 (Bahr and Peterson 2017). Of the rivers within the Carolinian Province that have population estimates (Table 34), the most recent population data for each river totals 3,069. Additionally, the Delaware River population is estimated at 12,047 adult shortnose sturgeon (95% CI = 10,757 – 13,580) (ERC

2006). The removal of 13 shortnose sturgeon in any 3 consecutive year period out of a population of 3,069 sturgeon in the Carolinian Province metapopulation (assumed to be an underestimate based on rivers with no population data) would result in the loss of 0.42% of the metapopulation and even less of the shortnose sturgeon population as a whole. This impact is even less significant to the species range wide considering the Delaware River population of shortnose is estimated at over 12,000 adults in that river alone.

We do not believe the loss of 14 individuals every 3 years will meaningful affect the reproductive potential of shortnose sturgeon range wide. If that loss was an adult female, the mortality could preclude the production of thousands of eggs, of which a fractional percentage would be expected to survive to sexual maturity. Thus, the death of a female would eliminate their contribution to future generations, and result in a reduction in reproduction. The loss of male may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Juveniles could also potentially account for the mortalities. We anticipate the overall impact to the population as whole from the loss of juveniles would generally be less, all things being equal, because they are generally more abundant than adults and are not yet sexually mature. Additionally, because of the requirements outlined in the Sturgeon PDCs the shortnose sturgeon spawning grounds themselves will not be affected by the proposed action.

The mortalities associated with the proposed action is not likely to reduce distribution of shortnose sturgeon. We anticipate a up to 2 individuals could be removed from the Savannah River over any consecutive 3-year period during cutterhead dredging, but we do not anticipate these potential removals would have any measurable effect on the overall distribution of the animals from the DPS. We anticipate the other mortalities occurring over any consecutive 3-year period could take place anywhere in the action area. Thus, we do anticipate any changes of the distribution of the species range wide.

Based on the information provided above, the nonlethal takes expected and the death of no more than 14 individuals during any consecutive 3-year period of the project, will not appreciably reduce the likelihood of shortnose sturgeon survival (i.e., they will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect shortnose sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent shortnose sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is because the death of 14 shortnose sturgeon represents an extremely small percentage of the range wide population; and those mortalities are also unlikely to affect genetic heterogeneity in the population; the loss of that individual is likely to have a small effect on reproductive output; and the action will have only a minor and temporary effect on the distribution of shortnose sturgeon in the action area and no effect on the distribution of the species range wide. Therefore, we do not believe the anticipated lethal takes will appreciably reduce the likelihood that shortnose sturgeon will survive in the wild.

8.6.2 Recovery

A Final Recovery Plan for the Shortnose Sturgeon was issued in December 1998 (NMFS 1998). The recovery objective and criteria are *"To recover populations to levels of abundance at which they no longer require protection under the ESA. For each population segment, the minimum population size will be large enough to maintain genetic diversity and avoid extinction."* The plan lists the following relevant actions needed for recovery:

> Objective 2: Protect Shortnose Sturgeon and their Habitats. Specifically, to mitigate impacts of modifications to important habitat and other destructive activities. Dredging is discussed in the Recovery Plan as a threat by entrainment and destroying benthic feeding areas, disrupting spawning migrations, and filling spawning habitat with resuspended fine sediments. Potential impacts from hydraulic dredge operations may be avoided by imposing work restrictions during sensitive time periods (i.e., spawning, migration, feeding) when sturgeon are most vulnerable to mortalities from dredging activity.

While the proposed action includes dredging, which is considered one of the major threats to shortnose sturgeon, we believe the PDCs in this Opinion will sufficiently minimize the effects of dredging on shortnose sturgeon. Specifically, the Sturgeon PDCs in Appendix E are designed to protect sturgeon aggregation areas to reduce take and harm to the species from being forced to relocate from a biologically important areas. The PDCs also consider water quality issues by identifying areas and times when rivers have had historically low DO and limit activities that would reduce the DO more. Last, the PDCs, protect spawning habitat by ensuring work occurs downstream of spawning habitat. We accordingly do not believe the proposed action will affect the recovery of shortnose sturgeon, by exacerbating effects of any of the major threats. The lethal take of 14 (8 observed and 6 unobserved) and nonlethal take of 6 shortnose sturgeon over any consecutive 3-year period of the project is not likely to appreciably reduce population numbers of any metapopulation over time due to current population sizes and expected recruitment. Therefore, we conclude the proposed action will not appreciably reduce the likelihood of recovery for shortnose sturgeon.

8.6.3 Conclusion

The lethal take of 14 (8 observed and 6 unobserved) and nonlethal take of 6 shortnose sturgeon associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of shortnose sturgeon in the wild.

8.7 Giant Manta Ray

The proposed action may result in the take of 89 giant manta rays over any consecutive 3-year period. We expect all takes to be nonlethal.

8.7.1 Survival

The proposed action may result in the nonlethal take of 89 giant manta rays during any consecutive 3-year period. The potential nonlethal capture of giant manta rays is not expected to have any measurable impact on the reproduction, numbers, or distribution of the species since all captures will be nonlethal, and animals are expected to be returned to the marine environment from the net without significant stress or injury, within the vicinity of where they are caught.

8.7.2 Recovery

A recovery plan for giant manta ray has not yet been developed; however, NMFS published a recovery outline for the giant manta ray (NMFS 2019a). The recovery outline identifies two primary interim goals: 1) to stabilize population trends through reduction of threats, such that the species is no longer declining throughout a significant portion of its range; and 2) to gather additional information through research and monitoring on the species' current distribution and abundance, movement and habitat use of adult and juveniles, mortality rates in commercial fisheries (including at-vessel and post-release mortality), and other potential threats that may contribute to the species' decline. The recovery outline serves as an interim guidance to direct recovery efforts for giant manta ray. An increase in the population to a size that maintains a steady recruitment of individuals representing all life stages would provide population stability and enable the population to sustain itself even in the event of unforeseen and unavoidable impacts. The major threats affecting the giant manta ray were summarized in the proposed rule (82 FR 3694, Publication Date January 12, 2017) and the final listing rule (83 FR 2619, Publication Date January 22, 2018), which stated that the most significant threat to the giant manta ray are overutilization by foreign commercial and artisanal fisheries in the Indo-Pacific and Eastern Pacific and inadequate regulatory mechanisms in foreign nations to protect these manta rays from the heavy fishing pressure and related mortality in these waters outside of U.S. jurisdiction. Other threats to M. birostris that potentially contribute to long-term risk of the species include (micro) plastic ingestion rates, increased parasitic loads as a result of climate change effects, and potential disruption of important life history functions as a result of increased tourism; however, due to the significant data gaps, the likelihood and impact of these threats on the status of the species is highly uncertain. None of the activities in this Opinion include activities that are considered threats to this species and we do not believe the proposed action will appreciably reduce the recovery of any Atlantic sturgeon DPS, by significantly exacerbating effects of any of the major threats identified in the proposed or final listing rules.

The individuals suffering nonlethal injuries or stresses are expected to fully recover such that no reductions in reproduction or numbers of giant manta rays are anticipated. The captures will occur at discrete locations where projects covered under this Opinion and the overall SARBO action area encompasses only a portion of the overall range/distribution of giant manta rays. Any incidentally caught animal would be released within the general area where caught and no change in the distribution of giant manta rays would be anticipated. Therefore, the nonlethal take of giant manta rays associated with the proposed action are not expected to cause an appreciable reduction in the likelihood of recovery of the giant manta rays in the wild.

8.7.3 Conclusion

The nonlethal take of 89 giant manta ray associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of giant manta ray in the wild.

8.8 Smalltooth sawfish (U.S. DPS)

The proposed action may result in the nonlethal take of 1 smalltooth sawfish over any consecutive 3-year period and 1 lethal take of a sawfish captured over any consecutive 9-year period, resulting from relocation trawling. The lethal capture accounts for smalltooth sawfish injured during relocation trawling that is assumed to result in an unobserved post-interaction mortality, as described in Section 6.1.4.4.

8.8.1 Survival

The nonlethal capture of 1 smalltooth sawfish over any consecutive 3-year period is not expected to have any measurable impact on the reproduction, numbers, or distribution of this species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of this species are anticipated. Since these captures may occur at any of the discrete project areas within the SARBO action area and would be released within the general area where caught, no change in the distribution of smalltooth sawfish is anticipated.

The loss of 1 smalltooth sawfish over any consecutive 9-year period will reduce the number of smalltooth sawfish as compared to the number of smalltooth sawfish that would have been present in the absence of the proposed action assuming all other variables remained the same. These lethal takes could also result in the loss of reproduction value as compared to the reproductive value in the absence of the proposed action, if a female is taken. An adult female smalltooth sawfish may have a litter of approximately 10 pups probably every 2 years; therefore, the loss of one adult female smalltooth sawfish, could preclude the production of these pups. Because smalltooth sawfish produce more well-developed young it is likely that some portion of these pups would have survived. Thus, the death of a female eliminates an individual's contribution to future generations, and the proposed action would result in a reduction in future smalltooth sawfish reproduction. The loss of 1 animal from the population over any consecutive 9-year period will have no impact on the distribution of the species.

While there is currently no accurate smalltooth sawfish population estimate, a trend analysis of their abundance in the Everglades National Park, considered within the species core range, shows a slightly increasing population abundance trend from 1972 - 2007 (Carlson et al. 2007). From 1989-2004, smalltooth sawfish relative abundance has increased 5% annually (Carlson and Osborne 2012a; NMFS 2010e). Using a demographic approach and life history data from similar species, Simpfendorfer (2000) estimates the most likely range for the intrinsic rate of increase is 0.08 per year to 0.13 per year with population doubling times of 10.3 to 13.5 years. Although this rate of population growth is very slow, the lethal take of 1 adult smalltooth sawfish over any consecutive 9-year period is not expected to have any measureable impact on this rate of population doubling-time and we do not believe these mortalities will have any measurable effect on these trends. Therefore, we believe the anticipated lethal and nonlethal take of smalltooth sawfish associated with the proposed action are not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the species in the wild.

Although we believe no change in distribution of smalltooth sawfish will occur as a result of the proposed action, we concluded the lethal take would result in an instantaneous reduction in absolute population numbers that may also reduce reproduction, but the short-term reductions are not expected to appreciably reduce the likelihood of survival of the species in the wild.

8.8.2 Recovery

The following analysis considers the effects of that take on the likelihood of recovery in the wild. We consider the recovery objectives in the recovery plan prepared for the species that relate to population numbers or reproduction that may be affected by the predicted reductions in the numbers or reproduction of smalltooth sawfish resulting from the proposed action. The recovery plan for the smalltooth sawfish (NMFS 2009c) lists 3 main objectives as recovery criteria for the species. The 2 objectives and the associated sub-objectives relevant to the proposed action are:

Objective - Minimize Human Interactions and Associated Injury and Mortality

Sub-objective:

- Minimize human interactions and resulting injury and mortality of smalltooth sawfish through public education and outreach targeted at groups that are most likely to interact with sawfish (e.g., fishermen, divers, boaters).
- Develop and seek adoption of guidelines for safe handling and release of smalltooth sawfish to reduce injury and mortality associated with fishing.
- Minimize injury and mortality in all commercial and recreational fisheries.

Objective - Ensure Smalltooth Sawfish Abundance Increases Substantially and the Species Reoccupies Areas from Which it had Previously Been Extirpated

Sub-objective:

- Sufficient numbers of juvenile smalltooth sawfish inhabit several nursery areas across a diverse geographic area to ensure survivorship and growth and to protect against the negative effects of stochastic events within parts of their range.
- Adult smalltooth sawfish (> 340 cm) are distributed throughout the historic core of the species’ range (both the Gulf of Mexico and Atlantic coasts of Florida). Numbers of adult smalltooth sawfish in both the Atlantic Ocean and Gulf of Mexico are sufficiently large that there is no significant risk of extirpation (i.e., local extinction) on either coast.
- Historic occurrence and/or seasonal migration of adult smalltooth sawfish are reestablished or maintained both along the Florida peninsula into the South-Atlantic Bight, and west of Florida into the northern and/or western Gulf of Mexico.

NMFS is currently funding several actions identified in the Recovery Plan for smalltooth sawfish; adult satellite tagging studies, the NSED, and monitoring take in commercial fisheries. Additionally, NMFS has developed safe-handling guidelines for the species. Despite the ongoing threats from activities which will be included as part of the proposed action, we have still seen a stable or slightly increasing trend in the status of this species. Thus, the proposed action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of the U.S. DPS of smalltooth sawfish's recovery in the wild. NMFS must continue to monitor the status of the population to ensure the species continues to recover.

Nonlethal takes of smalltooth sawfish will not affect the population of reproductive adult females. The potential lethal take of 1 smalltooth sawfish over any consecutive 9-year period will result in a reduction in overall population numbers in any given year. We have already determined that while these takes would likely result in an instantaneous reduction in absolute population numbers, we do not believe those reductions will have any measurably effects on the species increasing population trends. Additionally, we believe the proposed action will not impede the achievement of the relevant recovery objectives or sub-objectives. The SARBO action area does not occur in areas currently believed to be juvenile nursery areas. The loss of 1 smalltooth sawfish over any consecutive 9-year period is not likely to have any discernible effect on the distribution of smalltooth sawfish or the ability for the species to re-establish its historical occurrence or seasonal migrations. Thus, the effects of the proposed action will not result in an appreciable reduction in the likelihood of smalltooth sawfish recovery in the wild.

8.8.3 Conclusion

The lethal take of 1 smalltooth sawfish over any consecutive 9-year period and the nonlethal take of 1 smalltooth sawfish over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of smalltooth sawfish in the wild.

8.9 Johnson's Seagrass

The proposed action may result in the removal of up 6 acres of Johnson's seagrass during any consecutive 3-year period. Johnson's seagrass is a rhizome plant that spreads from a horizontal stem underground and cannot be meaningfully monitored by the loss of a single plant. Therefore, the loss of Johnson's seagrass will be tracked by the area of loss.

8.9.1 Survival

The loss of up to 6 acres of Johnson's seagrass annually will result in a reduction in numbers of the species; however, we believe that Johnson's seagrass is likely to recolonize areas where maintenance dredging occurs that are shallow enough to support this species and are located in areas with sufficient water clarity and quality. Johnson's seagrass is reported to effectively out-competes other seagrass species in periodically disturbed areas, based on its life history strategy (i.e., it) (Durako et al. 2003). The loss of up to 6 acres annually in areas of maintenance dredging covered under this Opinion will result in a reduction in the amount of Johnson's seagrass in those areas; however, continuing to maintain these channels is not expected to result in larger, population scale effects. As discussed above, seagrasses do occur in certain portions of the maintained channels, though these areas appear to recover over some period of time and seagrasses generally, and Johnson's seagrass specifically, are expected to recolonize between dredging cycles. We do not consider these repetitive, discrete losses of seagrasses as causing a threat to the survival of the species because these activities will not individually or cumulatively result in the long-term, large-scale mortality of Johnson's seagrass, particularly in light of its "pulsating patches" life history strategy, which allows the species to acclimate readily to disturbed areas. The loss of up to up to 6 acres annually of Johnson's seagrass will not result in long-term mortality either in the immediate action area of each project or on a larger scale within the range of Johnson's seagrass, particularly in light of its "pulsating patches" life history strategy, which allows the species to acclimate readily to disturbed areas. We expect that, in most places where Johnson's seagrass is present and disturbed, it will re-colonize and persist even though cyclical impacts have occurred. Pre and post-construction surveys have demonstrated that seagrass within these areas quickly recolonizes following the maintenance dredging events. Thus, the loss of Johnson's seagrass in the maintained channel is likely an overestimate since the estimated loss will be based on a survey for any seagrass and at least a portion of the seagrass loss will be seagrasses other than Johnson's seagrass. Since the listing in 1998, there have been no observed reductions in the species' geographic extent. However, the St. Johns River Water Management District (SJRWMD) observed H. johnsonii 21 km north of Sebastian Inlet on the western shore of the Indian River Lagoon; a discovery that slightly extended the species' previously known range limit (Virnstein and Hall 2009).

Reproduction will be temporarily reduced at each project site that reduces Johnson's seagrass numbers, but NMFS does not believe that this reproductive loss appreciably reduces the likelihood of survival of Johnson's seagrass in the wild. Johnson's seagrass will continue to reproduce and spread within the action area, even in those areas where it has been adversely affected, because the proposed adverse impacts are expected to be largely temporary (i.e., Johnson's seagrass is likely to recolonize the disturbed area and persist in other areas of the action area after a project is complete).

The proposed actions will not result in an appreciable reduction of Johnson's seagrass distribution or fragmentation of the range since we expect Johnson's seagrass will recolonize the disturbed areas and will continue to be capable of spreading via asexual fragmentation after the completion activities covered under this Opinion since the location of these activities are expected to be spread out over all of Johnson's seagrass range and critical habitat. Therefore, the reproductive potential of the species in the action area will persist.

8.9.2 Recovery

Recovery for Johnson's seagrass, as described in the recovery plan (NMFS 2002b), will be achieved when the recovery objectives are met (listed below). We review the project's expected impacts on critical habitat to determine whether it will be able to continue to provide its intended functions in achieving these recovery objectives.

> Objective: The species' present geographic range remains stable for at least 10 years or increases.

> Objective: Self-sustaining populations are present throughout the range at distances less than or equal to the maximum dispersal distance to allow for stable vegetative recruitment and genetic diversity.
>
> Objective: Populations and supporting habitat in its geographic range have long-term protection (through regulatory action or purchase acquisition).

The first recovery criterion for Johnson's seagrass is for its present range to remain stable for 10 years or to increase during that time. NMFS's 5-year review (2007a) of the status of the species concluded that the first recovery objective had been achieved as of 2007. In fact, the range had increased slightly northward at that time, and we have no information indicating range stability has decreased since then. We believe that the loss of up to 6 acres annually of Johnson's seagrass from activities covered under this Opinion will occur throughout the range of Johnson's seagrass and will not impede this recovery objective. These effects will not reduce or destabilize the present range of Johnson's seagrass. The loss of Johnson's seagrass from each of these activities is expected to be small and occur in individual, non-connected areas. In the case of the maintenance dredging, if Johnson's is present, it is because it has regrown in this area since the last dredging event and therefore is likely to recolonize this area again. Hence, these projects will not reduce the range of the species.

The second recovery criterion for Johnson's seagrass requires that self-sustaining populations be present throughout the range at distances less than or equal to the maximum dispersal distance for the species. Drifting fragments of Johnson's seagrass can remain viable in the water column for 4-8 days (Hall et al. 2006), and can travel several kilometers under the influence of wind, tides, and waves. Because of this, we believe that the removal of seagrasses from activities covered under this Opinion will not break up self-sustaining populations and that seagrass fragments will be able to drift to and over these impacted project sites. Therefore, we believe the loss of Johnson's seagrass associated with the proposed actions will not impede the recovery criterion.

The final recovery criterion is for populations and supporting habitat in the geographic range of Johnson's seagrass to have long-term protection (through regulatory action or purchase acquisition). All areas where maintenance dredging will occur within the range of Johnson's seagrass is limited to areas previously dredged. Since new areas are not covered under this Opinion, activities covered under this Opinion will not reduce the total area available to Johnson's seagrass within its known range and will therefore not impede achieving the recovery objectives listed above.

NMFS believes that the proposed actions will not appreciably reduce the likelihood of recovery of Johnson's seagrass in the wild. NMFS's 5-year review (2007) of the status of the species concluded that the first recovery objective has been achieved. In fact, the range has increased slightly northward. The proposed actions will not impact the status of this objective. Self-sustaining populations are present throughout the range of the species. The species' overall reproductive capacity will be only minimally reduced by the reduction in Johnson's seagrass numbers and reproduction resulting from the proposed actions. The proposed actions will not lead to separation of self-sustaining Johnson's seagrass patches to an extent that might lead to adverse effects to 1 or more patches of the species. Similarly, the proposed actions are not likely reduce the availability of suitable habitat in which the species can spread/flow in the future. The proposed actions will not reduce or destabilize the present range of Johnson's seagrass. Therefore, the activities will not appreciably reduce the likelihood of recovery of Johnson's seagrass in the wild.

8.9.3 Conclusion

The removal of up 6 acres annually of Johnson's seagrass associated with activities covered under this Opinion over any consecutive 3-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of Johnson's seagrass in the wild.

8.10 Staghorn Coral

The proposed actions covered under this Opinion may result in the following take of staghorn corals over any consecutive 10-year period: Up to 1,300 coral colonies relocated with 1,105 non-lethal and 195 lethal take of coral colonies associated with post-relocation mortality.

The Coral PDCs in Appendix C allow for relocation of coral colonies from out of the path of potential mortality from beach nourishment and associated pipeline placement projects to appropriate reef habitat nearby. While relocation minimizes the risk of lethal take, the relocation is a non-lethal take, and some lethal take will still occur. All coral relocation covered under this Opinion is limited to Southeast Florida within the range of ESA-listed corals. This Opinion does not cover pipeline or beach nourishment projects in the U.S. Caribbean.

8.10.1 Survival

The nonlethal take of 1,105 staghorn coral colonies associated with relocation over any consecutive 10-year period is not expected to have any measureable impact on the reproduction, abundance, or distribution of the species. All coral colonies that are relocated and survive will remain in the same general area and will therefore not change the number or distribution of the species. We believe that relocation of staghorn coral colonies is not likely to impact the ability of surviving individuals (those considered non-lethal take) to reproduce. Relocation of these colonies will not diminish the ability of these colonies to reproduce via sexual or asexual reproduction and therefore will have no change in the reproduction of these corals.

The lethal take of up to 195 staghorn corals over any consecutive 10-year period resulting from post-relocation mortality will result in a reduction in the number of the species. However, this loss is not expected to appreciably reduce the population of this species. A high number of colonies are believed to be still in existence throughout the species' range. A report on the status and trends of Caribbean corals over the last century indicates that the frequency of reefs at which staghorn coral was described as the dominant coral has remained stable (Jackson et al. 2014), and Southeast Florida contained some of the last remaining robust thickets of staghorn coral (Vargas-Angel et al. 2003), which we believe are no longer present following Hurricane Irma. However, any reestablished thickets will not be removed under this Opinion, as it limits coral removal to existing beach nourishment and pipeline placement corridor. Additionally, over 120,000 colonies of staghorn coral have been outplanted from coral nurseries to over 140 Florida reefs over the last 10 years (NMFS, unpublished data) to supplement natural populations and have resulted in higher density at sites where population enhancement has occurred (Miller et al. 2016). Thus, the loss of 195 staghorn corals through post-relocation mortality is expected to reduce the number of staghorn coral colonies, but is not expected to appreciably reduce the population of this species.

The lethal take of up to 195 staghorn corals over any consecutive 10-year period resulting from post-relocation mortality is not expected to impact the species' current geographic range. These anticipated mortalities would result in a reduction in staghorn corals distribution only in a limited number of discrete project areas in South Florida since the Coral PDCs in Appendix C limit beach nourishment and associated pipeline projects where corals may be relocated to areas outside of the U.S. Caribbean, and this species is found throughout the wider Caribbean region. The potential mortality of staghorn coral would cause no noticeable change or fragmentation in the distribution of the species, either in Florida or the wider Caribbean since relocation will be limited to corals that have regrown within previously used pipeline corridors and those near beach nourishment projects that could suffer mortality, if NMFS determines that they could be covered by sedimentation. These projects do not alter new areas of reef habitat in South Florida where staghorn occur on the existing reef tracts.

The lethal take of 195 staghorn coral colonies over any consecutive 10-year period is not likely to cause a measurable reduction in reproduction. Although staghorn coral reproduce sexually though broadcast spawning of sperm and eggs into the water column, sexual recruitment of all ESA-listed coral species is limited or absent in most locations. Staghorn corals primarily reproduce through asexual fragmentation, which is the reattachment of broken branches to suitable substrate to form a new colony. The fast growth rates of staghorn coral and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events. The loss of up to 195 colonies is unlikely to have any measurable effect on the remaining colonies, or the ability of those colonies to reproduce. The potential loss of up to 195 colonies would cause no noticeable change in the distribution of the species and would not appreciably reduce the number of colonies available for fragmentation. Therefore, the loss of up to 195 staghorn coral colonies is not likely to cause a measurable reduction in the species' reproduction.

After analyzing the magnitude of the effects of the proposed action on staghorn coral abundance, distribution, and reproduction, we believe the proposed action will not cause an appreciable reduction in the likelihood of survival of this species in the wild because it will not result in large-scale mortality either in the immediate action area of the projects or on a larger scale within the range of staghorn coral.

8.10.2 Recovery

A *Final Recovery Plan for Staghorn and Elkhorn Corals* was published March 5, 2015 (80 FR 12146, Publication Date March 6, 2015). The recovery plan outlines a recovery strategy for the species: Staghorn coral populations should be large enough so that successfully reproducing individuals comprise numerous populations across the historical range of the species and are large enough to protect their genetic diversity and maintain their ecosystem functions. Threats to the species and their habitat must be sufficiently abated to ensure a high probability of survival into the future. The recovery plan established 3 recovery criteria associated with the objective of ensuring population viability and 7 recovery criteria associated with the objective of eliminating or sufficiently abating global, regional, and local threats that contribute to the species' status. The best available information indicates that all recovery objectives must be met for staghorn coral to achieve recovery. The most relevant criteria to the impacts expected from the proposed project include the 3 population-based criteria:

> Criterion 1: Abundance
> Thickets are present throughout approximately 5% of consolidated reef habitat in 5 to 20 m water depth within the forereef zone. Thickets are defined as either a) colonies ≥ 0.5 m diameter in size at a density of 1 colony per m^2 or b) live staghorn coral benthic cover of approximately 25%. Populations with these characteristics should be present throughout the range and maintained for 20 years.
>
> Criterion 2: Genotypic Diversity
> Maintain current overall average genotypic diversity (proportion of unique genotypes per number of colonies sampled) of approximately 0.5 across these species' range.
>
> Criterion 3: Recruitment
> Observe recruitment rates necessary to achieve Criteria 1 and 2 over approximately 20 years; and
>
> Observe effective sexual recruitment (i.e., establishment of new larval derived colonies and survival to sexual maturity) in each species' population across their geographic range.

The nonlethal take of 1,105 staghorn coral colonies associated with relocation over any consecutive 10-year period is not expected to have any measureable impact on the abundance (Criterion 1), genotypic diversity (Criterion 2), or recruitment (Criterion 3) of the species. All coral colonies that are relocated will remain in the same general area and will therefore not change the number, genotypic diversity, or distribution of the species. Relocation of these colonies will not impact the ability of these colonies to reproduce via sexual or asexual reproduction and, therefore, will have no change in the recruitment of these corals.

The lethal take of 195 staghorn coral colonies is not expected to appreciably reduce the population-based recovery criteria of staghorn corals. The 195 staghorn coral colonies that are anticipated to suffer post-relocation mortality will not affect the presence and distribution of thickets throughout their range (Criterion 1). Thickets are a prominent feature at only a few known locations, and populations appear to consist mostly of isolated colonies or small groups of colonies compared to the vast thickets once prominent throughout its range. Although thickets may still occur in southeast Florida (many were lost during recent hurricanes) and some colonies in the thickets may need to be relocated, we anticipate that lethal take from post-relocation mortality would be of isolated colonies and not affect the entire thicket. Relocation of staghorn corals will only occur from areas within existing pipeline or beach nourishment templates, where we do not expect large thickets to be present. Thus, while the number of colonies within a thicket may be reduced, we do not anticipate a reduction in the density or size of colonies that define a thicket or in the distribution of thickets throughout their range. Lethal take is also not expected to affect the overall genotypic diversity of the species since the number of genotypes potentially lost is small compared to the overall number of colonies present throughout the range (Criterion 2). As discussed above, post-relocation mortality may result in a small reduction in number of colonies and in reproduction. We believe that recruitment in staghorn corals is not limited by the current population size (i.e., number of gametes available from colonies of reproductive size), but rather by the limited availability of suitable habitat for larval settlement and fragment attachment. Therefore, the reduction in reproduction is expected to be minimal and will not appreciably affect sexual recruitment (Criterion 3).

For the reasons stated above, we conclude that the non-lethal take of 1,105 staghorn coral colonies and the lethal take of 195 staghorn coral colonies from relocation over any consecutive 10-year period will not have a discernable impact on the population abundance recovery criteria of staghorn corals. Therefore, we conclude that the take of these colonies is not likely to appreciably reduce the recovery of staghorn corals in the wild.

8.10.3 Conclusion

The nonlethal take of 1,105 staghorn coral colonies and the lethal take of 195 staghorn corals from relocation over any consecutive 10-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of staghorn corals in the wild.

8.11 Elkhorn Coral

The proposed actions covered under this Opinion may result in the following take of elkhorn corals over any consecutive 10-year period: Up to 3 coral colonies relocated, with up to 1 lethal and 2 non-lethal take of coral colonies associated with post-relocation mortality. Because corals are colonial animals, it is impossible to measure impacts at the individual polyp level. Therefore, we analyze impacts to staghorn coral on the basis of impacts to coral colonies.

The Coral PDCs in Appendix C allow for relocation of coral colonies from out of the path of potential mortality from beach nourishment and associated pipeline placement projects to appropriate reef habitat nearby. All coral relocation covered under this Opinion is limited to Southeast Florida within the range of ESA-listed corals since the Coral PDCs in Appendix C do not cover pipeline or beach nourishment projects in the U.S. Caribbean.

8.11.1 Survival

The nonlethal take of 2 elkhorn coral colony associated with relocation over any consecutive 10-year period is not expected to have any measureable impact on the reproduction, abundance, or distribution of the species. If the relocated coral colonies survive, they will remain in the same general area and will therefore not change the number or distribution of the species. We believe that relocation of these colonies is not likely to impact the ability of the colonies to reproduce via sexual or asexual reproduction and therefore will have no change in the reproduction of these corals.

If the 1 relocated elkhorn colony dies, the lethal take will result in a reduction in the number of the species. However, loss of 1 colony over any consecutive 10-year period is not expected to appreciably reduce the population of this species. A report on the status and trends of Caribbean corals over the last century indicates that cover of elkhorn coral has remained relatively stable at approximately 1% throughout the region since the large mortality events of the 1970s and 1980s (Jackson et al. 2014). Elkhorn coral was present on about 20% of reefs surveyed between 2000 and 2011 and was dominant on approximately 5-10% of hundreds of reef sites surveyed throughout the Caribbean during the 4 periods of 1990-1994, 1995-1999, 2000-2004, and 2005-2011 (Jackson et al. 2014). Thus, the potential loss of 1 elkhorn coral colony over any consecutive 10-year period is expected to result in a reduction in numbers, but is not expected to appreciably reduce the population of this species.

The lethal take of 1 elkhorn coral over any consecutive 10-year period is not expected to impact the species' current geographic range. The anticipated mortality would result in a reduction in elkhorn coral distribution only in a limited discrete project area in South Florida since the Coral PDCs in Appendix C limit beach nourishment and associated pipeline projects where corals may be relocated to areas outside of the U.S. Caribbean, and this species is found throughout the wider Caribbean region. The potential mortality of 1 elkhorn coral colony over any consecutive 10-year period would cause no noticeable change or fragmentation in the distribution of the species, either in Florida or the wider Caribbean since relocation is limited to corals that have regrown within previously used pipeline corridors and those near beach nourishment projects that could suffer mortality if determined that they could be covered by sedimentation. These projects do not alter new areas of reef habitat in South Florida where elkhorn coral occur on the existing reef tracts.

The lethal take of 1 elkhorn coral colony over any consecutive 10-year period is not likely to cause a measurable reduction in reproduction. Although elkhorn coral reproduces sexually though broadcast spawning of sperm and eggs into the water column, sexual recruitment of all ESA-listed coral species is limited or absent in most locations. Elkhorn coral primarily reproduces through asexual fragmentation, which is the reattachment of broken branches to suitable substrate to form a new colony. The fast growth rates of elkhorn coral and propensity for formation of clones through asexual fragmentation enables it to expand between rare events of sexual recruitment and increases its potential for local recovery from mortality events. The loss of 1 colony over any consecutive 10-year period is unlikely to have any measurable effect on the remaining colonies or the ability of those colonies to reproduce. The potential loss of up to 1 colony of elkhorn coral over any consecutive 10-year period would cause no noticeable change in the distribution of the species and would not appreciably reduce the number of colonies available for fragmentation. Therefore, the loss of 1 elkhorn coral colony over any consecutive 10-year period is not likely to cause a measurable reduction in the species' ability to reproduce.

After analyzing the magnitude of the effects of the proposed action on elkhorn coral abundance, distribution, and reproduction, we believe the proposed action will not cause an appreciable reduction in the likelihood of survival of this species in the wild because it will not result in large-scale mortality either in the immediate action area of the projects or on a larger scale within the range of elkhorn coral.

8.11.2 Recovery

A *Final Recovery Plan for Staghorn and Elkhorn Corals* was published March 5, 2015 (80 FR 12146, Publication Date March 6, 2015). The recovery plan outlines a recovery strategy for the species: Elkhorn coral populations should be large enough so that successfully reproducing individuals comprise numerous populations across the historical range of the species and are large enough to protect their genetic diversity and maintain their ecosystem functions. Threats to the species and their habitat must be sufficiently abated to ensure a high probability of survival into the future. The recovery plan established 3 recovery criteria associated with the objective of ensuring population viability and 7 recovery criteria associated with the objective of eliminating or sufficiently abating global, regional, and local threats that contribute to the species' status. The best available information indicates that all recovery objectives must be met for staghorn coral to achieve recovery. The most relevant criteria to the impacts expected from the proposed project include the 3 population-based criteria:

> Criterion 1: Abundance
> Thickets are present throughout approximately 10% of consolidated reef habitat in 1 to 5 m water depth within the forereef zone. Thickets are defined as either a) colonies ≥ 1 m diameter in size at a density of 0.25 colony per m^2 or b) live elkhorn coral benthic cover of approximately 60%. Populations with these characteristics should be present throughout the range and maintained for 20 years.
>
> Criterion 2: Genotypic Diversity
> Maintain current overall average genotypic diversity (proportion of unique genotypes per number of colonies sampled) of approximately 0.5 across these species' range.
>
> Criterion 3: Recruitment
> Observe recruitment rates necessary to achieve Criteria 1 and 2 over approximately 20 years; and
>
> Observe effective sexual recruitment (i.e., establishment of new larval derived colonies and survival to sexual maturity) in each species' population across their geographic range.

The nonlethal take of 2 elkhorn coral colonies associated with relocation over any consecutive 10-year period is not expected to have any measureable impact on the abundance (Criterion 1), genotypic diversity (Criterion 2), or recruitment (Criterion 3) of the species. The relocated elkhorn coral colonies will remain in the same general area and will therefore not change the abundance, genotypic diversity, or range of the species. Relocation of these colonies will not impact its ability to reproduce via sexual or asexual reproduction and, therefore, will have no change in the recruitment of these corals.

The actions covered under this Opinion are not expected to appreciably affect the population-based recovery criteria of elkhorn coral. The 1 elkhorn coral colony that is anticipated to suffer post-relocation mortality over any consecutive 10-year period will not affect the presence and distribution of thickets throughout their range (Criterion 1). A thicket is a group of colonies defined by density and colony size. Because more than 1 colony is needed to form a thicket, the loss of 1 elkhorn colony from post-relocation mortality would not affect the distribution of thickets throughout their range. Lethal take is also not expected to affect the overall genotypic diversity of the species since the number of genotypes potentially lost is small compared to the overall number of colonies present throughout the range (Criterion 2). As discussed above, post-relocation mortality may result in a small reduction in number of colonies and in reproduction. We believe that recruitment in elkhorn coral is not limited by the current population size (i.e., number of gametes available from colonies of reproductive size), but rather by the limited availability of suitable habitat for larval settlement and fragment attachment. Therefore, the reduction in reproduction is expected to be minimal and will not appreciably affect sexual recruitment (Criterion 3).

For the reasons stated above, we conclude that the lethal take of 1 elkhorn coral colony and the non-lethal take of 2 elkhorn colonies relocated over any consecutive 10-year period will not have a discernable impact on the population abundance recovery criteria of elkhorn coral. Therefore, we conclude that the take associated with this Opinion is not likely to appreciably reduce the recovery of this species in the wild.

8.11.3 Conclusion

The lethal take of 1 elkhorn coral colony and non-lethal take of 2 elkhorn colonies from relocation over any consecutive 10-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of elkhorn coral in the wild.

8.12 Boulder Star, Lobed Star, and Mountainous Star Corals

The proposed actions covered under this Opinion may result in the following take of star corals over any consecutive 10-year period:

- Boulder star coral: Up to 74 coral colonies relocated with 63 non-lethal and 11 lethal take of coral colonies associated with post-relocation mortality.
- Lobed star coral: Up to 51 coral colonies relocated with 43 non-lethal and 8 lethal take of coral colonies associated with post-relocation mortality.
- Mountainous star coral: Up to 161 coral colonies relocated with 136 non-lethal and 25 lethal take of coral colonies associated with post-relocation mortality.

The Coral PDCs in Appendix C allow for relocation of coral colonies from out of the path of potential mortality from beach nourishment and associated pipeline placement projects to appropriate reef habitat nearby. All coral relocation covered under this Opinion is limited to Southeast Florida within the range of ESA-listed corals since the Coral PDCs in Appendix C do not cover pipeline or beach nourishment projects in the U.S. Caribbean. The Jeopardy analysis for the star corals (boulder star, lobed star, and mountainous star corals), which were listed at the same time and have the same recovery objectives, are analyzed together in this section. Long-term monitoring data sets and previous ecological studies did not distinguish among these 3 species and lumped them into to the *Orbicella* species complex. Distinguishing among the species is difficult, particularly for smaller colonies when differences in morphology are not evident. Therefore, much of the information available is for the complex as a whole (Brainard et al. 2011).

8.12.1 Survival

The nonlethal take of 63 boulder star, 43 lobed star, and 136 mountainous star coral colonies associated with relocation of corals covered over any consecutive 10-year period is not expected to have any measureable impact on the reproduction, abundance, or distribution of these species. All coral colonies that are relocated will remain in the same general area and will therefore not change the numbers or distribution of the species. Corals relocated and considered non-lethal take are expected to recover in the new location and be able to continue to be able to sexually reproduce at the same rate as before relocation. Therefore, the surviving relocated corals will have no change in the reproduction of these corals.

The lethal take of 11 boulder star, 8 lobed star, and 25 mountainous star coral colonies over any consecutive 10-year period will result in a reduction in the number of colonies available to contribute to reproduction of these species. While there is ample evidence that these species have declined dramatically throughout their range, a high number of colonies are believed to be still in existence through the species' range. However, the *Orbicella* species complex has historically been dominant on Caribbean and Florida coral reefs, characterizing the so-called "buttress zone" and "annularis zone" in the classical descriptions of Caribbean reefs (Goreau 1959). The listing rule noted that boulder star, lobed star, and mountainous star coral remain common and are often among the most abundant species on the reef. Therefore, we believe that, even with the recent declines discussed in Section 4.1.6.4, there are still high numbers of these corals throughout their range. Additionally, the lethal take of these species would result in a reduction in distribution only in a limited number of discrete project areas in South Florida. The Coral PDCs in Appendix C limit beach nourishment and associated pipeline projects where corals may be relocated to areas outside of the U.S. Caribbean, and these species are found throughout the wider Caribbean region. Relocation will be limited to corals that have regrown within previously used pipeline corridors and those near beach nourishment projects that could suffer mortality if determined that they could be covered by sedimentation. Projects under this Opinion will not alter new areas of reef habitat in South Florida. As compared to the range-wide population estimates, the potential loss associated with this action would cause no appreciable change in the population abundance or distribution of the species.

Boulder star, lobed star, and mountainous star corals reproduce by broadcast spawning sperm and eggs into the water column. Despite the common occurrence of these species on reefs and their high output of gametes, sexual recruits are rarely observed. We believe low recruitment rates are due to lack of suitable habitat available for larval settlement and are not caused by any shortage of gametes being released by reproductive colonies surviving in the wild. Therefore, the lethal take of 11 boulder star, 8 lobed star, and 25 mountainous star coral colonies over any consecutive 10-year period is not expected to appreciably reduce reproduction in these species.

After analyzing the magnitude of the effects of the proposed action on boulder star, lobed star, and mountainous star coral abundance, distribution, and reproduction, we believe the proposed action will not cause an appreciable reduction in the likelihood of survival of these 3 species in the wild because it will not result in large-scale mortality either in the immediate action area of the projects or on a larger scale within the range of these coral species.

8.12.2 Recovery

Boulder star, lobed star, and mountainous star corals do not currently have a recovery plan, but they have a Recovery Outline (NMFS 2015c) that guides recovery until a recovery plan is completed. The recovery vision statement in the Recovery Outline for boulder star coral, lobed star coral, and mountainous star coral states that populations of these species should be present across the historical range, with populations large enough and genetically diverse enough to support successful reproduction and recovery from mortality events and dense enough to maintain ecosystem function.

The nonlethal take of 63 boulder star, 43 lobed star, and 136 mountainous star coral colonies over any consecutive 10-year period associated with relocation of corals covered over any consecutive 10-year period is not expected to have a measureable impact on the reproduction, abundance, or genetic diversity of these species. All coral colonies that are relocated will remain in the same general area and will therefore not change the numbers, genetic diversity, or distribution of the species. Corals relocated and considered non-lethal take are expected to recover in the new location and be able to continue to be able to sexually reproduce at the same rate as before relocation. Therefore, the surviving relocated corals will have no change in the recovery ability of these corals.

The lethal take of 11 boulder star, 8 lobed star, and 25 mountainous star coral over any consecutive 10-year period through post-relocation mortality will not result in a discernable reduction in abundance, genotypic diversity, or reproduction of these species. As mentioned above the loss of these colonies is small in comparison to the range-wide population abundance and genetic diversity. We believe that reproduction in these species of coral is not limited by the current population size (i.e., number of gametes available from colonies of reproductive size), but rather by the limited availability of suitable habitat for larval settlement and recruitment. As stated above, we do not believe that the proposed action will appreciably reduce the reproduction of these species, due to the low level of sexual recruitment observed. Lethal take as a result of the proposed action will not result in large-scale mortality either in the immediate action area of the projects or on a larger scale within the range of these coral species, which would be expected to appreciably reduce the distribution of the species, or the genetic diversity of the species. Additionally, these species' are widely distributed in South Florida and the U.S. Caribbean, and the proposed action will only result in lethal take of individual colonies that have regrown within existing pipeline or beach nourishment templates. Therefore, we conclude that post-relocation mortality of these colonies is not likely to appreciably reduce the recovery of boulder star, lobed star, or mountainous star coral in the wild.

8.12.3 Conclusion

The nonlethal take of 63 boulder star, 43 lobed star, and 136 mountainous star coral colonies and the lethal take of 11 boulder star, 8 lobed star, and 25 mountainous star coral colonies from relocation over any consecutive 10-year period is not expected to cause an appreciable reduction in the likelihood of either the survival or recovery of elkhorn or staghorn corals in the wild.

9 CONCLUSION

NMFS analyzed the best available data, the status of the species, environmental baseline, effects of the proposed action, and cumulative effects to determine whether the proposed action is likely to jeopardize the continued existence of the following species.

Sea Turtles
The proposed action is not expected to appreciably reduce the likelihood of survival or recovery of these species in the wild. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of, NA DPS green, SA DPS green, Kemp's ridley, leatherback, and the NWA DPS loggerhead sea turtles.

Sturgeon
The proposed action is not expected to appreciably reduce the likelihood of survival or recovery of the 5 DPSs of Atlantic sturgeon and shortnose sturgeon. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and SA DPSs of Atlantic sturgeon, or shortnose sturgeon.

Giant Manta Ray
The proposed action is not expected to appreciably reduce the likelihood of survival or recovery of this species in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of the giant manta ray.

Smalltooth Sawfish
The proposed action is not expected to appreciably reduce the likelihood of survival or recovery of this species in the wild. Therefore, it is NMFS' Opinion that the proposed action is not likely to jeopardize the continued existence of the U.S. DPS of smalltooth sawfish.

Johnson's seagrass
The proposed action is not expected to appreciably reduce the likelihood of survival or recovery of this species in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of Johnson's seagrass.

Coral
The proposed action is not expected to appreciably reduce the likelihood of survival or recovery of these species in the wild. Therefore, it is NMFS' Opinion the proposed action is not likely to jeopardize the continued existence of elkhorn, staghorn, lobed star, mountainous star, or boulder star coral.

10 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption.

Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. *Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that would otherwise be considered prohibited under Section 9 or Section 4(d), but which is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the RPMs and the terms and conditions of the ITS of the Opinion.

The take of giant manta ray and ESA-listed corals by the proposed action is not prohibited, as no Section 4(d) Rules for the species have been promulgated. However, at least one circuit court case has held that non-prohibited incidental take must be included in the ITS.[82] Providing an exemption from Section 9 liability is not the only important purpose of specifying take in an ITS. Specifying incidental take ensures we have a metric against which we can measure whether or not reinitiation of consultation is required. It also ensures that we identify reasonable and prudent measures we believe are necessary or appropriate to minimize the impact of such incidental take.

Section 7(b)(4)(C) of the ESA specifies that to provide an ITS for an endangered or threatened species of marine mammal, the taking must be authorized under Section 101(a)(5) of the MMPA. No incidental take of marine mammals is anticipated as a result of the proposed action, and no incidental take of listed marine mammals has been authorized under Section 101(a)(5) of the MMPA. Therefore, no statement on incidental take of protected marine mammals is provided and no take is authorized. Nevertheless, the USACE and/or BOEM must immediately notify (within 24 hours, if communication is possible) NMFS' Office of Protected Resources if a take of a listed marine mammal occurs.

10.1 Anticipated Incidental Take

NMFS anticipates the following incidental takes of sea turtles, sturgeon, giant manta ray, smalltooth sawfish, and corals may occur in the future because of the proposed action.[83]
The level of take occurring annually is highly variable and influenced by sea temperatures, species abundances, and other factors that cannot be predicted. Because of this variability, it is unlikely that all species evaluated in this Opinion will be consistently impacted year after year. For example, some years may have no observed interactions and thus no estimated take. As a result, monitoring take using 1-year estimated take levels is largely impractical. For these reasons, and based on our experience monitoring fisheries, we believe a 3-year time period is appropriate for meaningful monitoring for sea turtles and sturgeon that are expected to have take occur every year. The triennial takes are set as 3-year running sums (total for any 3-year period)

[82] *CBD v. Salazar*, 695 F.3d 893 (9th Cir. 2012). Though the *Salazar* case is not a binding precedent for this action outside of the 9th Circuit, NMFS Southeast Region finds the reasoning persuasive and is following the case out of an abundance of caution and anticipation the ruling will be more broadly followed in future cases.
[83] Johnson's seagrass is not included in the Incidental Take Statement, as Incidental Take Statements are not provided for plants, only for fish and wildlife species USFWS and NMFS. 1998. Endangered Species Act consultation handbook. Procedures for Conducting Section 7 Consultations and Conferences. U.S. Fish and Wildlife, National Marine Fisheries Service.. *See, e.g., Ctr. for Biological Diversity v. Bureau of Land Mgmt.,* 833 F.3d 1136, 1145 (9th Cir. 2016)

and not for static 3-year periods (i.e., 2018-2021, 2019-2022, 2020-2023 and so on, as opposed to 2017-2020, 2021-2024, 2025-2028, etc.). This approach will allow us to incorporate the inherent variability in take levels, but still allow for an accurate assessment of how the proposed action is performing versus our expectations. For species that are estimated to take occur less frequently (i.e., ESA-listed coral, smalltooth sawfish, and giant manta ray), the take is estimated by a longer period of time provided in Table 52, but is still estimated on a consecutive number of years to account for variability in species presence and projects completed. These limits set thresholds that, if exceeded, would be the basis for reinitiating consultation.

Table 52. Anticipated Future Take Per 3 Consecutive Year Period

Species	Nonlethal Take-Observed	Lethal Take-Observed	Lethal Take-Unobserved	Total Lethal Observed + Unobserved Take	Sea Turtle Lost Egg Clutch
Green Sea Turtle NA DPS	742	59	59	118	3
Green Sea Turtle SA DPS	40	4	4	8	0
Kemp's Ridley Sea Turtle	1,340	58	58	116	1
Leatherback Sea Turtle	369	0	4	4	6
Loggerhead Sea Turtle NWA DPS	5,270	107	107	214	65
Atlantic Sturgeon South Atlantic DPS	499	73	0	73	N/A
Atlantic Sturgeon Carolina DPS	319	47	0	47	N/A
Atlantic sturgeon Chesapeake Bay DPS	91	14	0	14	N/A
Atlantic Sturgeon New York Bight DPS	34	5	0	5	N/A
Atlantic Sturgeon Gulf of Maine DPS	1	1	0	1	N/A
Shortnose sturgeon	6	8	6	14	N/A
Giant manta ray	89	0	0	0	N/A

Table 53. Anticipated Future Take Per Other Defined Time Period

Species	Nonlethal Take- Observed	Lethal Take- Observed	Lethal Take- Unobserved
Smalltooth sawfish (U.S. DPS)	1 total per 3 year period	1 total per 9 year period[84]	
Elkhorn Coral	2 total per 10 year period	1 total per 10 year period	Monitoring required = no unobserved
Staghorn coral	1,105 total per 10 year period	195 total per 10 year period	Monitoring required = no unobserved
Lobed star coral	43 total per 10 year period	8 total per 10 year period	Monitoring required = no unobserved
Mountainous star coral	136 total per 10 year period	25 total per 10 year period	Monitoring required = no unobserved
Boulder star coral	63 total per 10 year period	11 total per 10 year period	Monitoring required = no unobserved

[84] For smalltooth sawfish, a total of 3 takes in authorized every 9 years, with up to 1 lethal take.

10.2 Effect of the Take

NMFS has determined the level of anticipated take specified in Section 10.1 of this Opinion is not likely to jeopardize the continued existence of the following ESA-listed species or DPS: Northwest Atlantic DPS of loggerhead sea turtles, green sea turtles (South and North Atlantic DPSs), leatherback sea turtles, Kemp's ridley sea turtles, smalltooth sawfish (U.S. DPS), Atlantic sturgeon (the Gulf of Maine, Chesapeake, New York Bight, Carolina and the SA DPSs), shortnose sturgeon, giant manta ray, smalltooth sawfish, Johnson's seagrass, and corals (elkhorn, staghorn, lobed star, mountainous star, and boulder star corals).

10.3 Reasonable and Prudent Measures

Section 7(b)(4) of the ESA requires NMFS to issue a statement specifying the impact of any incidental take on listed species, which results from an agency action otherwise found to comply with Section 7(a)(2) of the ESA. It also states that the RPMs necessary to minimize the impacts of take and the terms and conditions to implement those measures must be provided and must be followed to minimize those impacts. Only incidental taking by the federal agency or applicant that complies with the specified terms and conditions is consistent with this Opinion and authorized.

The RPMs and terms and conditions are specified as required by 50 CFR 402.14 (i)(1)(ii) and (iv) to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. These measures and terms and conditions are nondiscretionary, and must be implemented by the USACE and/or BOEM or the applicants (in regulatory actions) in order for the protection of Section 7(o)(2) to apply. The USACE and/or BOEM has a continuing duty to regulate the activity covered by this ITS. If the USACE and/or BOEM or the applicants fail to adhere to the terms and conditions of the ITS through enforceable terms, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse. To monitor the impact of the incidental take, the USACE and/or BOEM or the applicant must report the progress of the action and its impact on the species to NMFS as specified in the ITS [50 CFR 402.14(i)(3)]. All reports required under this Opinion will be sent to **SERO.Dredge@noaa.gov**.

The PDCs of this Opinion are designed to minimize take including those provided to increase the ability to observe take by hopper dredging in the General PDCs in Appendix B and the PSO PDCs in Appendix H. The PDCs area also designed to minimize or remove the risk of take of ESA-listed corals (Appendix C), Johnson's seagrass (Appendix D), Atlantic and shortnose sturgeon (Appendix E), and North Atlantic right whales (Appendix F). In light of these protections, NMFS concludes that the PDCs include the measures necessary and appropriate to minimize the impact of incidental take. Therefore, we include as RPMs/T&Cs those elements of the proposed action that are necessary and appropriate to minimize the impact of take. The RPMs/T&Cs impose no additional requirements beyond those specified by the proposed action. Only incidental takes that occur while these measures are in full implementation are authorized. These restrictions remain valid until reinitiation and conclusion of any subsequent Section 7 consultation.

NMFS has identified the following measures that must be implemented by the USACE and BOEM (directly or through mandatory conditions of its authorization for the action) to minimize the impact of take from the proposed action.

1. The USACE and BOEM will have measures in place to minimize and avoid interactions with any protected species resulting from the proposed action, as appropriate.
2. Relocation trawling is authorized and will be used in reasonable circumstances to reduce lethal take from hopper dredging.
3. Relocation of ESA-listed corals will be used as a minimization measure to reduce lethal take from pipeline placement and beach nourishment.
4. All handling of ESA-listed species will be monitored for and handled by a NMFS-approved PSO.

10.4 Terms and Conditions

In order to be exempt from liability for take prohibited by Section 9 of the ESA, the USACE and/or BOEM or their applicants must comply with all of the conditions of this Opinion outlined in the proposed action and PDCs, specifically those required to minimize or prevent incidental take outlined in the RPMs in Section 10.3 of this Opinion.

1. All equipment will be operated according to the requirements in the PDCs relating to direct efforts to monitor, minimize, or avoid impacts on ESA-listed species, including those applicable: generally to projects covered by SARBO 2020 (Appendix B), by equipment type (Appendix B and Appendix G), and by location (Appendix C for projects within the range of ESA-listed corals and Appendix E for projects in sturgeon rivers).
2. All personnel associated with projects authorized under the 2020 SARBO will be educated regarding the requirements to avoid and minimize effects to ESA-listed species and critical habitat, consistent with Appendix B (RPM 1).
3. Reporting requirements necessary to document take of ESA-listed species will be met by following the reporting requirements outlined in Section 2.9 of this Opinion (RPM 1).
4. Relocation trawling will be conducted according to the PDC requirements in Appendix B and Appendix I (RPM 2).
5. Relocation of corals will be done if NMFS determines that relocation would minimize the impact of lethal take, following a project-specific review. Relocation of corals will be done in accordance with the requirements of the PDCs, in Appendix C (RPM3).
6. A PSO will monitor for the presence of ESA-listed species on hopper dredges and relocation trawling vessels and will be responsible for handling, tagging, collecting genetic samples, and recording the details of the capture in accordance with Appendix B, Appendix H, and Section 2.9 of this Opinion (RPM 4).

11 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. In order for the Service to be kept informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, the Service requests notification of the implementation of any conservation recommendations.

Data Collection

1. NMFS recommends the USACE and BOEM consider making the data collected as part of any required surveys (field, species, habitat, etc.) as well as required monitoring and reporting available to the public and scientific community in an easily accessible online database (such as ODESS) that can be queried to aggregate data across required reports. Access to such data will help to better understand the biology of listed species (e.g., their range), as well as inform future consultations and authorizations by providing information on the effectiveness of the conservation measures and the impact of dredging activity on listed species.

Species Monitoring

2. NMFS recommends the USACE and BOEM work with members of the Florida Atlantic Coast Telemetry and Atlantic Cooperative Telemetry Networks to identify opportunities to fill voids in the existing receiver arrays along the east coast of the United States. Arrays are useful in monitoring a variety of species including listed smalltooth sawfish, sea turtles, giant manta rays, and sturgeon.

Dredge Equipment and Species Interactions

3. NMFS recommends the USACE and BOEM should explore alternative means for monitoring interactions with listed species when MEC/ UXO screening is in place. This could include exploring the potential for video or other electronic monitoring and consider designing pilot studies to test the efficiency of innovative monitoring and screening techniques.
4. NMFS recommends the USACE and BOEM conduct studies to evaluate differences in species take by different hopper dredge designs to determine whether some designs may result in higher likelihood of take. This should include an evaluation of the design of both the hopper dredging and the draghead deflector shield and options to minimize take in challenging locations such as areas with high debris and uneven bottom surfaces.
5. NMFS recommends the USACE and BOEM evaluate the feasibility of installing video or other remote-sensing equipment (e.g., GoPro) on the dragarm or draghead to determine whether visibility is sufficient to monitor for interactions with species. If installing such equipment is feasible, and visibility is sufficient to observe and identify species encounters, the USACE should design a study to test species reactions to the dredge or the disturbance radius from the hopper dredge draghead.

6. NMFS recommends the USACE and BOEM examine different inflow and overflow screening and box configurations to minimize the risk of clogging during dredging while maximizing the ability for PSOs to easily, and safely inspect all of the contents collected in the boxes for evidence of take. Possibilities include different placement within the hopper dredge that are more easily viewable, various box sizes and shapes that improve visibility without entering the box, and various screening designs and materials that reduce clogging. Improved technological solutions such as video monitoring capability may also prove useful in reducing the need for PSOs to enter the boxes to inspect the contents.
7. NMFS recommends the USACE and BOEM continue to support the development of innovative new dredging methods/practices and dredge designs that will further minimize listed species interactions and mortalities. This could include a study to observe listed species reactions to dredging to understand how the species react to the oncoming draghead (e.g., disturbance radius, behavioral response) in different conditions (e.g., bottom topography, temperature).
8. NMFS recommends the USACE and BOEM develop standard procedures to remove marine debris excavated during dredging operations. Marine debris creates an entanglement risk and pose risk to listed species when consumed. Standard procedures should be developed and implemented by action agencies to necessitate surface marine debris removal during dredging operations.
9. NMFS recommends the USACE and BOEM conduct or support research that evaluates known, commonly used biomarkers for physiological stress (e.g., stress hormone levels) or other sublethal impacts of listed species taken during relocation activities. This information could help us better determine the condition of listed species post release and more accurately assess post-release mortality that will inform future consultations.
10. NMFS recommends the USACE and BOEM explore the aggregate impacts of their activities through the development of Population Consequence of Disturbance models for listed species. Population Consequence of Disturbance models simulate the cumulative effects of sublethal stressors across individuals to characterize the population consequences of anthropogenic activities including sound exposure, pollutants, and reduced habitat access. The Population Consequence of Disturbance modeling framework typically uses a bioenergetic model as a transfer function between stressors (e.g., behavioral disturbance) and their impacts on vital rates (i.e., growth, reproduction).
11. NMFS recommends the USACE and BOEM designs pilot studies and support literature searches to parameterize bioenergetic models for listed species. We recommend that the USACE or BOEM design pilot studies to develop dose-response function for modeling the effects of sublethal stressors (e.g., what is the probability of a behavioral response at different levels of sound exposure). This will support the development of Population Consequence of Disturbance models for listed species.
12. NMFS recommends the USACE and BOEM further examine hopper dredge designs currently in use to determine what features and practices could allow entrainment from a point not associated with the drag head. Past examples of occasional sea turtles and sturgeon found unharmed in the hopper indicates that some individuals may enter the hopper without having passed through the draghead.

13. NMFS recommends the USACE and BOEM consider testing the feasibility of innovative techniques (e.g., side scan sonar) to improve observing or identify if sturgeon, sea turtles, or other ESA-listed species are present in the path of dredging or trawling activities. If effective, results could identify times and locations when dredging or relocation trawling should or should not be used. This could reduce take if dredging in high density locations can be delayed to another time or reduce cost of relocation trawlers if the area has a low risk of species interaction.

North Atlantic right whale-specific recommendations

14. NMFS recommends the USACE and BOEM partner and participate in "table top" exercises that may be conducted to enhance contingency plans and preparedness for responding to river incursions by North Atlantic right whales.
15. NMFS recommends the USACE and BOEM work with North Atlantic right whale biologists or other related biologists from the states of Florida, Georgia, and South Carolina to identify suitable right whale necropsy sites, support site preparation, and obtain any appropriate environmental permits.
16. NMFS recommends the USACE and BOEM support development of minimally invasive tags and tracking studies to gain insights into North Atlantic right whale spatio-temporal movements as well as time spent at various locations within the water column.

Seagrass-specific recommendations

17. NMFS recommends the USACE consider using remote sensing data in a bio-optical model to predict seagrass coverage within the Intracoastal Waterway.
18. NMFS recommends the USACE conduct long-term water quality research to assist with the development baseline conditions within the Intracoastal Waterway, seagrass thresholds (specifically Johnson's seagrass), and the relationship between turbidity and sediment deposition on seagrass communities.
19. NMFS recommends the USACE conduct studies to better understand and predict the water quality and turbidity/sedimentation effects to seagrasses from different types of dredging removal and placement.

Sea turtle-specific recommendations

20. NMFS recommends that the USACE and BOEM conduct or support directed research to understand sea turtle use of and movement in, the water column in the summer. Warmer water temperatures, and breeding and nesting activities, likely result in different sea turtle behavior and movements within the water column compared to other times of year when hopper dredging occurs. Information on water column use during that time is important to understand the likelihood of sea turtle interactions with hopper dredges, and to inform hopper-dredging practices during the summer months.
21. NMFS recommends the USACE and BOEM conduct studies to determine the effectiveness of using a sea turtle deflector to minimize the potential entrainment of sturgeon during hopper dredging.

Sturgeon-specific recommendations

22. NMFS recommends the USACE and BOEM support or conduct a study that would compile and standardize existing telemetry data for sturgeon. These data could be used to create innovative video animations and mapping exercises that would inform species

distribution, seasonality, and possibly habitat use/preference. A better understanding of distribution, seasonality, and habitat use may also refine when and where consultations are needed, potentially reducing workloads. These data will also inform future consultations and authorizations by providing information on the effectiveness of PDCs.

23. NMFS recommends the USACE consider contacting the University of Delaware – College of Earth, Ocean, and Environment regarding their Real-Time Sturgeon Predictive modeling. The potential expansion of this existing model may provide USACE with insight into when sturgeon interactions with dredge equipment are more probable. This increased awareness may allow the USACE to more effectively deploy relocation trawlers or alter their dredge plans to reduce impacts to sturgeon.

Coral-specific recommendations

24. NMFS recommends the USACE and BOEM provide the location and size of all non-listed corals within areas that may be impacted to all persons who hold the proper permits and who may be interested in rescuing those corals for use in research or educational activities.

25. NMFS recommends the USACE and BOEM conduct interstitial pore water bacterial composition of dredge materials to determine if the sediment contains potentially pathogenic bacteria that is harmful to coral and may become mobilized during dredging operations.

26. NMFS recommends the USACE and BOEM conduct long-term water quality research to assist with the development of coral water quality thresholds based on the relationship between turbidity generated during dredging or material placement and sediment deposition on coral reefs.

27. NMFS recommends the USACE and BOEM gather long-term baseline hydrodynamic data to feed into sediment transport modeling. This would make the models more accurate and lead to increased confidence when predicting sediment fate and transport from dredge projects to listed coral species.

Giant manta ray-specific recommendations

28. NMFS recommends the USACE and BOEM conduct studies or support directed research to satellite (SPOT 6; Mini PAT) or acoustic tag giant manta rays in the action area. Data collected from tagging would be used evaluate residency and diel movement patterns, and purported nearshore nursery habitat along Florida east coast, which will inform future consultation and authorizations.

29. NMFS recommends the USACE and BOEM require all personnel to report giant manta ray sightings to the giant manta ray recovery coordinator at NMFS Southeast Region Protected Resources Division. Giant manta ray’s observations should be photographed and include the latitude/longitude, date, and environmental conditions at the time of the sighting.

12 REINITIATION OF CONSULTATION

This concludes NMFS's formal consultation on the proposed action. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal action agency involvement or control over the action has been retained, or is authorized by law, and if (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action on listed species or designated critical habitat in a manner or to an extent not considered in this Opinion, (3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this Opinion, or (4) a new species is listed or critical habitat designated that may be affected by the action.

13 LITERATURE CITED

Abal, E. G., and coauthors. 1994. Physiological and morphological responses of the seagrass *Zostera capricorni* Aschers, to light intensity. Journal of Experimental Marine Biology and Ecology 178(1):113-129.

Ackerman, R. A. 1997. The nest environment and embryonic development of sea turtles. Pages 83-106 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Acropora Biological Review Team. 2005. Atlantic *Acropora* Status Review Document. Biological Review Team, Report to National Marine Fisheries Service, Southeast Regional Office.

Adams, D. H., and E. Amesbury. 1998. Occurrence of the manta ray, *Manta birostris*, in the Indian River Lagoon, Florida. Florida Scientist 61(1):7-9.

Addison, D. S. 1997. Sea turtle nesting on Cay Sal, Bahamas, recorded June 2-4, 1996. Bahamas Journal of Science 5(1):34-35.

Addison, D. S., and B. Morford. 1996. Sea turtle nesting activity on the Cay Sal Bank, Bahamas. Bahamas Journal of Science 3(3):31-36.

Adey, W. H. 1978. Coral reef morphogenesis: A multidimensional model. Science 202(4370):831-837.

Aguilar, R., J. Mas, and X. Pastor. 1995. Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle *Caretta caretta* population in the western Mediterranean. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-361, Miami, FL.

Aguirre, A. A., G. H. Balazs, T. R. Spraker, S. K. K. Murakawa, and B. Zimmerman. 2002. Pathology of oropharyngeal fibropapillomatosis in green turtles *Chelonia mydas*. Journal of Aquatic Animal Health 14:298-304.

Al-Bahry, S. N., and coauthors. 2009. Ultrastructural features and elemental distribution in eggshell during pre and post hatching periods in the green turtle, *Chelonia mydas* at Ras Al-Hadd, Oman. Tissue and Cell 41(3):214-221.

Alcolado, P. M., and coauthors. 2010. Condition of remote reefs off southwest Cuba. Ciencias Marinas 36(2):179-197.

Allen, A. S., A. C. Seymour, and D. Rittschof. 2017. Chemoreception drives plastic consumption in a hard coral. Marine Pollution Bulletin 124(1):198-205.

Anan, Y., T. Kunito, I. Watanabe, H. Sakai, and S. Tanabe. 2001. Trace element accumulation in hawksbill turtles (*Eretmochelys imbricata*) and green turtles (*Chelonia mydas*) from Yaeyama Islands, Japan. Environmental Toxicology and Chemistry 20(12):2802-2814.

ANSI. 1986. Methods of measurement for impulse noise. Acoustical Society of America, ANSI S12.7-1986, New York, NY.

ANSI. 1995. Bioacoustical terminology. Acoustical Society of America, ANSI S3.20-1995, New York, NY.

Antonelis, G. A., J. D. Baker, T. C. Johanos, R. C. Braun, and A. L. Harting. 2006. Hawaiian monk seal (Monachus schauinslandi): status and conservation issues. Atoll Research Bulletin 543:75-101.

Arendt, M., and coauthors. 2017. Temporal and spatial distribution of Atlantic sturgeon (*Acipenser oxyrinchus*) in U.S. territorial waters off South Carolina and Georgia: Final (2013-2017) report to the National Marine Fisheries Service, National Oceanic and Atmospheric Administration. South Carolina Department of Natural Resources, Marine Resources Division and Georgia Department of Natural Resources, Grant Number NA13NMF4720045, Charleston, SC.

Arendt, M. D., and coauthors. 2009. Examination of local movement and migratory behavior of sea turtles during spring and summer along the Atlantic Coast off the Southeastern United States. South Carolina Department of Natural Resources, Grant Number NA03NMF4720281.

Armstrong, J. L., and J. E. Hightower. 2002. Potential for restoration of the Roanoke River population of Atlantic sturgeon. Journal of Applied Ichthyology 18(4-6):475-480

Aronson, R. B., and W. F. Precht. 2001. White-band disease and the changing face of Caribbean coral reefs. Hydrobiologia 460(1-3):25-38.

ASMFC. 2017. Atlantic Sturgeon Benchmark Stock Assessment and Peer Review Report. Atlantic States Marine Fisheries Commission, Arlington, VA.

Atlantic States Marine Fisheries Commission. 1998. American shad and Atlantic sturgeon stock assessment peer review: Terms of reference and advisory report. Atlantic States Marine Fisheries Commission, NOAA Award No. NA87FG0025, Washington, D.C.

Atlantic States Marine Fisheries Commission. 2007. Special report to the Atlantic Sturgeon Management Board: Estimation of Atlantic sturgeon bycatch in coastal Atlantic commercial fisheries of New England and the Mid-Atlantic. Atlantic States Marine Fisheries Commission, Arlington, VA.

Atlantic States Marine Fisheries Commission. 2011. 69th annual report of the Atlantic States Marine Fisheries Commission. Atlantic States Marine Fisheries Commission, Arlington, VA.

Atlantic States Marine Fisheries Commission. 2017. 2017 Atlantic Sturgeon Benchmark Stock Assessment and Peer Review Report. Atlantic States Marine Fisheries Commission, NOAA Award No.NA15NMF4740069, Arlington, VA.

Atlantic Sturgeon Status Review Team. 1998. Status Review of Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

Atlantic Sturgeon Status Review Team. 2007. Status review of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

Avens, L., J. C. Taylor, L. R. Goshe, T. T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8:165-177.

Babcock, R., and L. Smith. 2000. Effects of sedimentation on coral settlement and survivorship, volume 1, Bali, Indonesia.

Bahn, R. A., J. E. Fleming, and D. L. Peterson. 2012. Bycatch of shortnose sturgeon in the commercial American shad fishery of the Altamaha River, Georgia. North American Journal of Fisheries Management 32(3):557-562.

Bahr, D. L., and D. L. Peterson. 2016. Recruitment of juvenile Atlantic sturgeon in the Savannah River, Georgia Transactions of the American Fisheries Society 145(6):1171-1178.

Bahr, D. L., and D. L. Peterson. 2017. Status of the shortnose sturgeon population in the Savannah River, Georgia. Transactions of the American Fisheries Society 146(1):92-98.

Bain, M. B. 1997. Atlantic and shortnose sturgeons of the Hudson River: Common and divergent life history attributes. Environmental Biology of Fishes 48(1-4):347-358.

Bain, M. B., N. Haley, D. L. Peterson, J. R. Waldman, and K. Arend. 2000. Harvest and habitats of Atlantic sturgeon *Acipenser oxyrinchus* Mitchill, 1815 in the Hudson River estuary: Lessons for sturgeon conservation. Boletín - Instituto Español de Oceanografía 16(1-4):43-53.

Baird, R. W., and coauthors. 2015. False killer whales and fisheries interactions in Hawaiian waters: Evidence for sex bias and variation among populations and social groups. Marine Mammal Science 31(2):579-590.

Bak, R. P. M. 1977. Coral reefs and their zonation in Netherlands Antilles. AAPG Studies in Geology 4:3–16.

Bak, R. P. M. 1978. Lethal and sublethal effects of dredging on reef corals. Marine Pollution Bulletin 9(1):14-16.

Bak, R. P. M., and S. R. Criens. 1982. Experimental fusion in Atlantic *Acropora* (Scleractinia). Marine Biology Letters 3:67-72.

Baker, J. D., C. L. Littnan, and D. W. Johnston. 2006. Potential effects of sea-level rise on terrestrial habitat and biota of the Northwestern Hawaiian Islands. Endangered Species Research 2:21-30.

Balazik, M. T., D. J. Farrae, T. L. Darden, and G. C. Garman. 2017. Genetic differentiation of spring-spawning and fall-spawning male Atlantic sturgeon in the James River, Virginia. PLOS ONE 12(7):8.

Balazik, M. T., G. C. Garman, J. P. Van Eenennaam, J. Mohler, and L. C. Woods III. 2012a. Empirical evidence of fall spawning by Atlantic sturgeon in the James River, Virginia. Transactions of the American Fisheries Society 141(6):1465-1471.

Balazik, M. T., and J. A. Musick. 2015. Dual annual spawning races in Atlantic sturgeon. PLOS ONE 10(5):16.

Balazik, M. T., and coauthors. 2012b. The potential for vessel interactions with adult Atlantic sturgeon in the James River, Virginia. North American Journal of Fisheries Management 32(6):1062-1069.

Balazs, G. H. 1979. Growth rates of immature green turtles in the Hawaiian Archipelago. Smithsonian Institution Press, Washington, D.C.

Balazs, G. H. 1983. Recovery records of adult green turtles observed or originally tagged at French Frigate Shoals, northwestern Hawaiian Islands. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center, NOAA Technical Memorandum NMFS-SWFC-36 and UNIHI-SEAGRANT-CR-83-03, Honolulu, HI.

Balazs, G. H. 1985. Impact of ocean debris on marine turtles: Entanglement and ingestion. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center, Technical Memorandum NMFS-SWFC-54, Honolulu, HI.

Barber, J. R., K. R. Crooks, and K. M. Fristrup. 2010. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology and Evolution 25(3):180-189.

Barber, N. L., and T. C. Stamey. 2000. Droughts in Georgia. U.S. Department of the Interior, U.S. Geological Survey, Open-File Report 00-380, Norcross, GA.

Barbieri, E. 2009. Concentration of heavy metals in tissues of green turtles (*Chelonia mydas*) sampled in the Cananéia Estuary, Brazil. Brazilian Journal of Oceanography 57(3):243-248.

Barnette, M. C. 2017. Potential impacts of artificial reef development on sea turtle conservation in Florida. U.S. Department of Commerce, National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Southeast Regional Office, NOAA Technical Memorandum NMFS-SER-5, Saint Petersburg, FL.

Bartol, S. M., J. A. Musick, and M. L. Lenhardt. 1999. Auditory evoked potentials of theloggerhead sea turtle (*Caretta caretta*). Copeia 1999(3):836-840.

Bass, A. L., C. J. Lagueux, and B. W. Bowen. 1998. Origin of green turtles, *Chelonia mydas*, at "Sleeping Rocks" off the northeast coast of Nicaragua. Copeia 1998(4):1064-1069.

Bass, A. L., and W. N. Witzell. 2000. Demographic composition of immature green turtles (*Chelonia mydas*) from the east central Florida coast: Evidence from mtDNA markers. Herpetologica 56(3):357-367.

Baughman, J. L. 1943. Notes on sawfish, *Pristis perotteti* Müller and Henle, not previously reported from the waters of the United States. Copeia 1943(1):43-48.

Baums, I. B., and coauthors. 2013. Genotypic variation influences reproductive success and thermal stress tolerance in the reef building coral, *Acropora palmata*. Coral Reefs 32:703-717.

Baums, I. B., C. R. Hughes, and M. E. Hellberg. 2005a. Mendelian microsatellite loci for the Caribbean coral *Acropora palmata*. Marine Ecology Progress Series 288:115-127.

Baums, I. B., M. E. Johnson, M. K. Devlin-Durante, and M. W. Miller. 2010. Host population genetic structure and zooxanthellae diversity of two reef-building coral species along the Florida Reef Tract and wider Caribbean. Coral Reefs 29:835–842.

Baums, I. B., M. W. Miller, and M. E. Hellberg. 2005b. Regionally isolated populations of an imperiled Caribbean coral, *Acropora palmata*. Molecular Ecology 14(5):1377-1390.

Baums, I. B., M. W. Miller, and M. E. Hellberg. 2006a. Geographic variation in clonal structure in a reef-building Caribbean coral, *Acropora palmata*. Ecological Monographs 76(4):503-519.

Baums, I. B., C. B. Paris, and L. M. Chérubin. 2006b. A bio-oceanographic filter to larval dispersal in a reef-building coral. Limnology and Oceanography 51(5):1969-1981.

Beauvais, S. L., S. B. Jones, S. K. Brewer, and E. E. Little. 2000. Physiological measures of neurotoxicity of diazinon and malathion to larval rainbow trout (*Oncorhynchus mykiss*) and their correlation with behavioral measures. Environmental Toxicology and Chemistry 19(7):1875-1880.

Becker, L. C., and E. Mueller. 2001. The culture, transplantation and storage of *Montastraea faveolata, Acropora cervicornis* and *Acropora palmata*: What we have learned so far. Bulletin of Marine Science 69(2):881-896.

Bejarano-Álvarez, M., F. Galván-Magaña, and R. I. Ochoa-Báez. 2011. Reproductive biology of the scalloped hammerhead shark *Sphyrna lewini* (Chondrichthyes: Sphyrnidae) off south-west Mexico. International Journal of Ichthyology 17(1):11-23.

Belovsky, G. E. 1987. Extinction models and mammalian persistence. Pages 35-58 *in* M. E. Soulé, editor. Viable Populations for Conservation. Cambridge University Press.

Benson, S. R., and coauthors. 2007a. Post-nesting migrations of leatherback turtles (*Dermochelys coriacea*) from Jamursba-Medi, Bird's Head Peninsula, Indonesia. Chelonian Conservation and Biology 6(1):150-154.

Benson, S. R., and coauthors. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere 2(7):27.

Benson, S. R., K. A. Forney, J. T. Harvey, J. V. Carretta, and P. H. Dutton. 2007b. Abundance, distribution, and habitat of leatherback turtles (*Dermochelys coriacea*) off California, 1990−2003. Fishery Bulletin 105(3):337-347.

Berlin, W. H., R. J. Hesselberg, and M. J. Mac. 1981. Growth and mortality of fry of Lake Michigan lake trout during chrinic exposure to PCB's and DDE. Pages 11-22 *in* Chlorinated Hydrocarbons as a Factor in the Reproduction and Survival of Lake Trout (*Salvelinus namaycush*) in Lake Michigan. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Washington, D.C.

Berry, R. J. 1971. Conservation and the genetical constitution of populations. Pages 177-206 *in* E. Duffey, and A. S. Watt, editors. The Scientific Management of Animal and Plant Communities for Conservation:The 11th Symposium of the British Ecological Society, University of East Anglia, Norwich, 7-9 July 1970. Blackwell Scientific Publications, Oxford, U.K.

Bessudo, S., and coauthors. 2011. Residency of the scalloped hammerhead shark (*Sphyrna lewini*) at Malpelo Island and evidence of migration to other islands in the Eastern Tropical Pacific. Environmental Biology of Fishes 91(2):165-176.

Bethea, D. M., K. L. Smith, and J. K. Carlson. 2012. Relative abundance and essential fish habitat studies for smalltooth sawfish, *Prisits pectinata*, in southwest Florida, USA. NOAA Fisheries Southeast Fisheries Science Center, Panama City, FL.

Bethea, D. M., K. L. Smith, L. D. Hollensead, and J. K. Carlson. 2011. Relative abundance and essential fish habitat studies for smalltooth sawfish, *Pristis pectinata*, in southwest Florida, USA. NOAA Fisheries, Southeast Fisheries Science Center, Panama City, FL.

Bigelow, H. B., and coauthors. 1963. Soft-rayed bony fishes, Class Osteichthyes, Order Acipenseroidei, Order Lepisostei, Order Isospondyli, Suborder Elopoidea, Suborder Clupeoidea, Suborder Salmonoidea. Pages 630 *in* H. B. Bigelow, editor. Fishes of the Western North Atlantic, Part 3, First edition. Sears Foundation for Marine Research, Yale University, New Haven, CT.

Bigelow, H. B., and W. C. Schroeder. 1953a. Fishes of the Gulf of Maine, 1st edition, volume 53. U.S. Government Printing Office, Washington, D.C.

Bigelow, H. B., and W. C. Schroeder. 1953b. Sawfishes, guitarfishes, skates, and rays. Chimaeroids. Pages 588 *in* J. Tee-Van, C. M. Breder, A. E. Parr, W. E. Schroeder, and L. P. Schultz, editors. Fishes of the Western North Atlantic, Part 2. Sears Foundation of Marine Research, Yale University, New Haven, CT.

Biggs, R. B. 1968. Environmental effects of overboard spoil disposal. Journal of the Sanitary Engineering Division 94(3):477-488.

Billsson, K., L. Westerlund, M. Tysklind, and P. E. Olsson. 1998. Developmental disturbances caused by polychlorinated biphenyls in zebrafish (*Brachydanio rerio*). Marine Environmental Research 46(1–5):461-464.

Birrell, C. L., L. J. McCook, and B. L. Willis. 2005. Effects of algal turfs and sediment on coral settlement. Marine Pollution Bulletin 51(1-4):408-14.

Bjorndal, K. A. 1982. The consequences of herbivory for the life history pattern of the Caribbean green turtle, *Chelonia mydas*. Pages 111-116 *in* K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles: Proceedings of the World Conference on Sea Turtle Conservation. Smithsonian Institution Press, Washington, D.C.

Bjorndal, K. A., and A. B. Bolten. 2008. Annual variation in source contributions to a mixed stock: Implications for quantifying connectivity. Molecular Ecology 17(9):2185-2193.

Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. 2005. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. Ecological Applications 15(1):304-314.

Bjorndal, K. A., A. B. Bolten, T. Dellinger, C. Delgado, and H. R. Martins. 2003. Compensatory growth in oceanic loggerhead sea turtles: Response to a stochastic environment. Ecology 84(5):1237-1249.

Bjorndal, K. A., J. A. Wetherall, A. B. Bolten, and J. A. Mortimer. 1999. Twenty-six years of green turtle nesting at Tortuguero, Costa-Rica: An encouraging trend. Conservation Biology 13(1):126-134.

Blair, H. B., N. D. Merchant, A. S. Friedlaender, D. N. Wiley, and S. E. Parks. 2016. Evidence for ship noise impacts on humpback whale foraging behaviour. Biology Letters 12(8).

Blunden, J., and D. S. Arndt (editors). 2016. State of the climate in 2015. Bulletin of the American Meteorological Society 97(8):300.

BOEM. 2014. Atlantic OCS proposed geological and geophysical activities Mid-Atlantic and South Atlantic planning areas final programmatic environmental impact statement. U.S. Department of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans.

BOEM. 2017. Review of Sea Turtle Entrainment Risk by Trailing Suction Hopper Dredges in the US Atlantic and Gulf of Mexico and the Development of the ASTER Decision Support Tool

BOEM. 2019. Sand survey activities for BOEM's Marine Minerals Program, Atlantic and Gulf of Mexico: Final environmental assessment. U.S. Department of the Interior, Bureau of Ocean Energy Management, Marine Minerals Program, OCIS EIS/EA BOEM 2019-022, Sterling, VA.

Bolam, S. G., and H. L. Rees. 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: A habitat approach. Environmental Management 32(2):171-188.

Bolker, B. M., T. Okuyama, K. A. Bjorndal, and A. B. Bolten. 2007. Incorporating multiple mixed stocks in mixed stock analysis: 'Many-to-many' analyses. Molecular Ecology 16(4):685-695.

Bolten, A. B., K. A. Bjorndal, and H. R. Martins. 1994. Life history model for the loggerhead sea turtle (*Caretta caretta*) population in the Atlantic: Potential impacts of a longline fishery. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOAA Technical Memorandum NMFS-SWFSC-201, Honolulu, HI.

Bolten, A. B., and coauthors. 1998. Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis. Ecological Applications 8(1):1-7.

Bolten, A. B., and B. E. Witherington. 2003. Loggerhead Sea Turtles. Smithsonian Books, Washington, D.C.

Boreman, J. 1997. Sensitivity of North American sturgeons and paddlefish to fishing mortality. Environmental Biology of Fishes 48(1-4):399-405.

Borodin, N. 1925. Biological observations on the Atlantic sturgeon (*Acipenser sturio*). Transactions of the American Fisheries Society 55(1):184-190.

Bostrom, B. L., and D. R. Jones. 2007. Exercise warms adult leatherback turtles. Comparative Biochemistry and Physiology A: Molecular and Integrated Physiology 147(2):323-31.

Bouchard, S., and coauthors. 1998. Effects of exposed pilings on sea turtle nesting activity at Melbourne Beach, Florida. Journal of Coastal Research 14(4):1343-1347.

Bowen, B. W., and coauthors. 1992. Global population structure and natural history of the green turtle (*Chelonia mydas*) in terms of matriarchal phylogeny. Evolution 46(4):865-881.

Bowlby, C. E., G. A. Green, and M. L. Bonnell. 1994. Observations of leatherback turtles offshore of Washington and Oregon. Northwestern Naturalist 75(1):33-35.

Boysen, K. A., and J. J. Hoover. 2009. Swimming performance of juvenile white sturgeon (*Acipenser transmontanus*): Training and the probability of entrainment due to dredging. Journal of Applied Ichthyology 25(s2):54-59.

Brainard, R. E., and coauthors. 2011. Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center, NOAA Technical Memorandum NMFS-PIFSC-27, Honolulu, HI.

Brame, A. B., and coauthors. 2019. Biology, ecology, and status of the smalltooth sawfish *Pristis pectinata* in the USA. Endangered Species Research 39:9-23.

Braun, C. D., G. B. Skomal, S. R. Thorrold, and M. L. Berumen. 2015. Movements of the reef manta ray (*Manta alfred*i) in the Red Sea using satellite and acoustic telemetry. Marine Biology 162(12):2351-2362.

Bresette, M. J., R. A. Scarpino, D. A. Singewald, and E. P. de Maye. 2006. Recruitment of post-pelagic green turtles (*Chelonia mydas*) to nearshore reefs on Florida's southeast coast. Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Bright, A. J., D. E. Williams, K. L. Kramer, and M. W. Miller. 2013. Recovery of *Acropora palmata* in Curacao: A comparison with the Florida Keys. Bulletin of Marine Science 89(3):747-757.

Brown, J. J., and G. W. Murphy. 2010. Atlantic Sturgeon Vessel-Strike Mortalities in the Delaware Estuary. Fisheries 35(2):72-83.

Bruckner, A. W. 2012. Factors contributing to the regional decline of *Montastraea annularis* (complex). James Cook University, Cairns, Australia.

Bruckner, A. W., and R. J. Bruckner. 2006. The recent decline of *Montastraea annularis* (complex) coral populations in western Curaçao: A cause for concern? Revista de biología tropical 54:45-58.

Bruckner, A. W., and R. L. Hill. 2009. Ten years of change to coral communities off Mona and Desecheo Islands, Puerto Rico, from disease and bleaching. Diseases of Aquatic Organisms 87(1-2):19-31.

Brundage, H. M., and J. C. O' Herron. 2003. Population estimate for shortnose sturgeon in the Delaware River.

Buckley, J., and B. Kynard. 1985. Habitat use and behavior of pre-spawning and spawning shortnose sturgeon, *Acipenser brevirostrum*, in the Connecticut River. Pages 111-117 *in* F. P. Binkowski, and S. I. Doroshov, editors. North American sturgeons. W. Junk Publishers, Dordrecht, Netherlands.

Budd, A. F., H. Fukami, N. D. Smith, and N. Knowlton. 2012. Taxonomic classification of the reef coral family Mussidae (Cnidaria: Anthozoa: Scleractinia). Zoological Journal of the Linnean Society 166(3):465-529.

Bulthuis, D. A. 1983. Effects of in situ light reduction on density and growth of the seagrass *Heterozostera tasmanica* (Martens ex Aschers.) den Hartog in Western Port, Victoria, Australia. Journal of Experimental Marine Biology and Ecology 67(1):91-103.

Burchfield, P. M. 2013. Gladys Porter Zoo's preliminary annual report on the Mexico/United States of America population restoration project for the Kemp’s ridley sea turtle, *Lepidochelys kempii*, on the coasts of Tamaulipas, Mexico 2013. Gladys Porter Zoo, Brownsville, TX.

Burgess, K. B., and coauthors. 2016. Manta birostris, predator of the deep? Insight into the diet of the giant manta ray through stable isotope analysis. Royal Society Open Science 3(11):160717.

Burton, W. H. 1993. Effects of bucket dredging on water quality in the Delaware River and the potential for effects on fisheries resources. Versar, Inc., Columbia, MD.

Bush, A. 2003. Diet and diel feeding periodicity of juvenile scalloped hammerhead sharks, *Sphyrna lewini*, in Kāne'ohe Bay, Ō'ahu, Hawai'i. Environmental Biology of Fishes 67(1):1-11.

Bushnoe, T. M., J. A. Musick, and D. S. Ha. 2005. Essential spawning and nursery habitat of Atlantic sturgeon (*Acipenser oxyrinchus*) in Virginia. Virginia Institute of Marine Science, Gloucester Point, VA.

Bythell, J. C. 1990. Nutrient uptake in the reef-building coral *Acropora palmata* at natural environmental concentrations. Marine Ecolology Progress Series 68:65-69.

Cairns, S. D. 1982. Stony corals (Cnidaria: Hydrozoa, Scleractinia) of Carrie Bow Cay, Belize. Pages 271-302 *in* K. Rützler, and I. G. Macintyre, editors. The Atlantic Barrier Reef Ecosystem at Carrie Bow Cay, Belize, volume I. Structure and Communities. Smithsonian Contributions to the Marine Sciences, Washington, D.C.

Caldwell, D. K., and A. Carr. 1957. Status of the sea turtle fishery in Florida. Wildlife Management Institute, Washington, D.C.

Caldwell, S. 1990. Texas sawfish: Which way did they go? Tide (Jan-Feb):16-19.

Cameron, P., J. Berg, V. Dethlefsen, and H. von Westernhagen. 1992. Developmental defects in pelagic embryos of several flatfish species in the Southern North Sea. Netherlands Journal of Sea Research 29(1–3):239-256.

Cameron, S. 2012. Assessing the impacts of channel dredging on Atlantic sturgeon movement and behavior. Presented to the Virginia Atlantic Sturgeon Partnership Meeting, Charles City, VA.

Campbell, J. G., and L. R. Goodman. 2004. Acute sensitivity of juvenile shortnose sturgeon to low dissolved oxygen concentrations. Transactions of the American Fisheries Society 133(3):772-776.

Campell, C. L., and C. J. Lagueux. 2005. Survival probability estimates for large juvenile and adult green turtles (*Chelonia mydas*) exposed to an artisanal marine turtle fishery in the western Caribbean. Herpetologica 61(2):91-103.

Carballo, J. L., C. Olabarria, and T. G. Osuna. 2002. Analysis of four macroalgal assemblages along the Pacific Mexican coast during and after the 1997-98 El Niño. Ecosystems 5(8):749-760.

Carlson, J. K., C. S. Horn, and S. B. Creager. 2019. Safe Handling and Release Guidelines for Manta and Devil Rays (Mobulid Species).

Carlson, J. K., and J. Osborne. 2012a. Relative abundance of smalltooth sawfish (*Pristis pectinata*) based on the Everglades National Park Creel Survey. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-626, Panama City, FL.

Carlson, J. K., and J. Osborne. 2012b. Relative abundance of smalltooth sawfish (*Pristis pectinata*) based on the Everglades National Park Creel Survey. NOAA National Marine Fisheries Service, NMFS-SEFSC-626, Panama City, FL.

Carlson, J. K., J. Osborne, and T. W. Schmidt. 2007. Monitoring the recovery of smalltooth sawfish, *Pristis pectinata*, using standardized relative indices of abundance. Biological Conservation 136(2):195-202.

Carlson, J. K., and C. A. Simpfendorfer. 2015. Recovery potential of smalltooth sawfish, Pristis pectinata, in the United States determined using population viability models. Aquatic Conservation: Marine and Freshwater Ecosystems 25(2):187-200.

Caron, F., D. Hatin, and R. Fortin. 2002. Biological characteristics of adult Atlantic sturgeon (*Acipenser oxyrinchus*) in the St Lawrence River estuary and the effectiveness of management rules. Journal of Applied Ichthyology 18(4-6):580-585.

Carpenter, R. C. 1986. Partitioning herbivory and its effects on coral reef algal communities. Ecological Monographs 56(4):345-363.

Carr, A. 1986. New perspectives on the pelagic stage of sea turtle development. Caribbean Conservation Corporation and U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center, Panama City Laboratory, NOAA Technical Memorandum NMFS-SEFSC-190, Panama City, FL.

Carr, T., and N. Carr. 1991. Surveys of the sea turtles of Angola. Biological Conservation 58(1):19-29.

Carricart-Ganivet, J. P., N. Cabanillas-Terán, I. Cruz-Ortega, and P. Blanchon. 2012. Sensitivity of calcification to thermal stress varies among genera of massive reef-building corals. PLOS ONE 7(3):e32859.

Casper, B. M. 2006. The hearing abilities of elasmobranch fishes. Dissertation. University of South Florida, Tampa, FL.

Casper, B. M., M. B. Halvorsen, and A. N. Popper. 2012. Are sharks even bothered by a noisy environment? Pages 93-97 *in* A. N. Popper, and A. Hawkins, editors. The Effects of Noise on Aquatic Life. Advances in Experimental Medicine and Biology, volume 730. Springer, New York, NY.

Casper, B. M., and D. A. Mann. 2007. Dipole hearing measurements in elasmobranch fishes. The Journal of experimental biology 210:75-81.

Caurant, F., P. Bustamante, M. Bordes, and P. Miramand. 1999. Bioaccumulation of cadmium, copper and zinc in some tissues of three species of marine turtles stranded along the French Atlantic coasts. Marine Pollution Bulletin 38(12):1085-1091.

Cech Jr., J. J., and S. I. Doroshov. 2005. Environmental requirements, preferences, and tolerance limits of North American sturgeons. Pages 73-86 *in* G. T. O. LeBreton, F. W. H. Beamish, and R. S. McKinley, editors. Sturgeons and Paddlefish of North America, volume 27. Kluwer Academic Publishers, New York, NY.

Chaloupka, M. 2002. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. Ecological Modelling 148(1):79-109.

Chaloupka, M. Y., and C. J. Limpus. 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Marine Biology 146(6):1251-1261.

Chaloupka, M. Y., and J. A. Musick. 1997. Age, growth, and population dynamics. Pages 233-276 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Chaloupka, M. Y., T. M. Work, G. H. Balazs, S. K. K. Murakawa, and R. Morris. 2008. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982-2003). Marine Biology 154(5):887-898.

Chassot, E., M. Amandè, C. Pierre, R. Pianet, and R. Dédo. 2008. Some preliminary results on tuna discards and bycatch in the French purse seine fishery of the eastern Atlantic Ocean. Collective Volume Of Scientific Papers 64.

Cheng, L., and coauthors. 2017. Improved estimates of ocean heat content from 1960 to 2015. Science Advances 3(3).

Chin, A., P. Kyne, T. Walker, and R. McAuley. 2010. An integrated risk assessment for climate change: Analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. Global Change Biology 16:1936-1953.

Chytalo, K. 1996. Summary of Long Island Sound dredging windows strategy workshop. Atlantic States Marine Fisheries Commission.

CITES. 2010. Consideration of proposals for amendment of Appendices I and II (CoP15 Prop. 15). 15th meeting of the Conference of the Parties, Doha (Qatar). Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

CITES. 2013. Consideration of proposals for amendment of Appendices I and II: Manta Rays. Convention on International Trace in Endangered Species of Wild Fauna and Flora (CITES), Sixteenth Meeting of the Conference of the Parties, CoP16 Prop. 46 (Rev. 2), Bangkok, Thailand.

Clark, T. B. 2010. Abundance, home range, and movement patterns of manta rays (*Manta alfredi, M. birostris*) in Hawai'i. Dissertation. Univeristy of Hawai'i at Mānoa, Honolulu, HI.

Clarke, J. U., C. J. McGrath, and T. J. Estes. 2011. Sampling strategies for Confined Disposal Facilities (CDF) characterization. U.S. Department of Defense, Army Corps of Engineers, Engineer Research and Development Center, Dredging Operations and Environmental Research (DOER), ERDC TN-DOER-D12, Vicksburg, MS.

Clausner, J. E., and D. R. Jones. 2004. Prediction of flow fields near the intakes of hydraulic dredges. U.S. Department of Defense, U.S. Army Corps of Engiineers, Engineer Research and Development Center, Dredging Operation and Environmental Research (DOER) Program, https://dots.el.erdc.dren.mil/doer/tools.html.

Clement, D. 2013. Effects on marine mammals. Cawthron Institute, Nelson, New Zealand.

Coastal Planning & Engineering. 2013. Final Biological Assessment, Broward County Shore Protection Project, Segment II. Coastal Planning & Engineering, Inc., Boca Raton, FL.

Colella, M. A., R. R. Ruzicka, J. A. Kidney, J. M. Morrison, and V. B. Brinkhuis. 2012. Cold-water event of January 2010 results in catastrophic benthic mortality on patch reefs in the Florida Keys. Coral Reefs 31:621-632.

Coles, R. J. 1916a. Natural history notes on the devil-fish, *Manta birostris* (Walbaum) and *Mobula olfersi* (Muller). Bulletin of the American Museum of Natural History 35(33):649-657.

Coles, R. J. 1916b. Notes on Radcliffe's sharks and rays of Beaufort. Copeia 32:45-47.

Collazo, J. A., R. Boulon Jr., and T. L. Tallevast. 1992. Abundance and growth patterns of *Chelonia mydas* in Culebra, Puerto Rico. Journal of Herpetology 26(3):293-300.

Collette, B. B., and G. Klein-MacPhee (editors). 2002. Bigelow and Schroeder's Fishes of the Gulf of Maine, 3rd edition. Smithsonian Institution Press, Washington, D.C.

Collins, M. R., and coauthors. 2003. Shortnose sturgeon in the Santee-Cooper reservoir system, South Carolina. Transactions of the American Fisheries Society 132(6):1244-1250.

Collins, M. R., W. C. Post, D. C. Russ, and T. I. J. Smith. 2002. Habitat use and movements of juvenile shortnose sturgeon in the Savannah River, Georgia-South Carolina. Transactions of the American Fisheries Society 131(5):975-979.

Collins, M. R., S. G. Rogers, T. I. J. Smith, and M. L. Moser. 2000a. Primary factors affecting sturgeon populations in the southeastern United States: Fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66(3):917-928.

Collins, M. R., and T. I. J. Smith. 1993. Characteristics of the adult segment of the Savannah River population of shortnose sturgeon. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 47:485-491.

Collins, M. R., and T. I. J. Smith. 1997. Management Briefs: Distributions of shortnose and Atlantic sturgeons in South Carolina. North American Journal of Fisheries Management 17(4):955-1000.

Collins, M. R., T. I. J. Smith, W. C. Post, and O. Pashuk. 2000b. Habitat utilization and biological characteristics of adult Atlantic sturgeon in two South Carolina rivers. Transactions of the American Fisheries Society 129(4):982-988.

Compagno, L. J. V. 1984. FAO Species Catalogue. Vol. 4. Sharks of the World. An annotated and illustrated catalogue of shark species known to date. Part 2. Carcharhiniformes. Food and Agriculture Organization of the United Nations, Fisheries Synopsis No. 125, Rome, Italy.

Conant, T. A., and coauthors. 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the Loggerhead Biological Review Team to the National Marine Fisheries Service, August 2009.

Consortium, I. R. L. 2015. 2011 Superbloom Report: Evaluating effects and possible causes with available data.

Convention on Migratory Species. 2014. Proposal for the inclusion of the reef manta ray (*Manta alfredi*) in CMS Appendix I and II. Convention on Migratory Species (CMS), 18th Meeting of the Scientic Council, UNEP/CMS/ScC18/Doc.7.2.9, Bonn, Germany.

Cooke, D. W., J. P. Kirk, J. J. V. Morrow, and S. D. Leach. 2004. Population dynamics of a migration limited shortnose sturgeon population. Pages 82-91 *in* Annual Conference, Southeastern Association of Fish and Wildlife Agencies.

Cooper, K. 1989. Effects of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans on aquatic organisms. Reviews in Aquatic Sciences 1(2):227-242.

Cortés, E. 1999. Standardized diet compositions and trophic levels of sharks. ICES Journal of Marine Science 56(5):707-717.

Couturier, L. I. E., and coauthors. 2012. Biology, ecology and conservation of the Mobulidae. Journal of Fish Biology 80(5):1075-1119.

Couturier, L. I. E., and coauthors. 2013. Stable isotope and signature fatty acid analyses suggest reef manta rays feed on demersal zooplankton. PLOS ONE 8(10):e77152.

Crance, J. H. 1987. Habitat suitability index curves for anadromous fishes. Pages 554 *in* M. J. Dadswell, and coeditors, editors. Common Strategies of Anadromous and Catadromous Fishes. American Fisheries Society Symposium I. American Fisheries Society, Bethesda, MD.

Crocker, C. E., and J. J. Cech Jr. 1997. Effects of environmental hypoxia on oxygen consumption rate and swimming activity in juvenile white sturgeon, Acipenser transmontanus, in relation to temperature and life intervals. Environmental Biology of Fishes 50:383-289.

Crouse, D. T. 1999. Population modeling and implications for Caribbean hawksbill sea turtle management. Chelonian Conservation and Biology 3(2):185-188.

Crouse, D. T., L. B. Crowder, and H. Caswell. 1987. A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology 68(5):1412-1423.

Crowder, L. B., D. T. Crouse, S. S. Heppell, and T. H. Martin. 1994. Predicting the impact of turtle excluder devices on loggerhead sea turtle populations. Ecological Applications 4(3):437-445.

Cruz-Piñón, G., J. P. Carricart-Ganivet, and J. Espinoza-Avalos. 2003. Monthly skeletal extension rates of the hermatypic corals *Montastraea annularis* and *Montastraea faveolata*: Biological and environmental controls. Marine Biology 143(3):491-500.

Cunning, R., R. N. Silverstein, B. B. Barnes, and A. C. Baker. 2019. Extensive coral mortality and critical habitat loss following dredging and their association with remotely-sensed sediment plumes. Marine Pollution Bulletin 145:195-199.

D'Ilio, S., D. Mattei, M. F. Blasi, A. Alimonti, and S. Bogialli. 2011. The occurrence of chemical elements and POPs in loggerhead turtles (*Caretta caretta*): An overview. Marine Pollution Bulletin 62(8):1606-1615.

Dadswell, M. J. 1979. Biology and population characteristics of the shortnose sturgeon, *Acipenser brevirostrum* LeSueur 1818 (Osteichthyes: Acipenseridae), in the Saint John River Estuary, New Brunswick, Canada. Canadian Journal of Zoology 57:2186-2210.

Dadswell, M. J. 2006. A review of the status of Atlantic sturgeon in Canada, with comparisons to populations in the United States and Europe. Fisheries 31(5):218-229.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984. Synopsis of biological data on shortnose sturgeon, *Acipenser brevirostrum* LeSueur 1818. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NMFS 14, Silver Spring, Maryland.

Dahl, T. E., and C. E. Johnson. 1991. Status and trends of wetlands in the conterminous United States, mid-1970s to mid-1980s. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Washington, D.C.

Dalton, T., and D. Jin. 2010. Extent and frequency of vessel oil spills in US marine protected areas. Marine Pollution Bulletin 60(11):1939-1945.

Damon-Randall, K., M. Colligan, and J. Crocker. 2013. Composition of Atlantic sturgeon in rivers, estuaries and in marine waters. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, Gloucester, MA.

Daniels, R. C., T. W. White, and K. K. Chapman. 1993. Sea-level rise: Destruction of threatened and endangered species habitat in South Carolina. Environmental Management 17(3):373-385.

Davenport, J., D. L. Holland, and J. East. 1990a. Thermal and biochemical characteristics of the lipids of the leatherback turtle (*Dermochelys coriacea*): Evidence of endothermy. Journal of the Marine Biological Association of the United Kingdom 70:33-41.

Davenport, J., J. Wrench, J. McEvoy, and V. Carnacho-Ibar. 1990b. Metal and PCB concentrations in the "Harlech" leatherback. Marine Turtle Newsletter 48:1-6.

Davis, G. E. 1982. A century of natural change in coral distribution at the Dry Tortugas: A comparison of reef maps from 1881 and 1976. Bulletin of Marine Science 32(2):608-623.

Dawes, C. J., C. S. Lobban, and D. A. Tomasko. 1989. A comparison of the physiological ecology of the seagrasses *Halophila decipiens* Ostenfeld and *H. johnsonii* Eiseman from Florida. Aquatic Botany 33(1-2):149-154.

Deakos, M. H. 2010. Ecology and social behavior of a resident manta ray (*Manta alfredi*) population of Maui, Hawai'i. Dissertation. Univeristy of Hawai'i at Mānoa, Honolulu, HI.

Deakos, M. H., J. D. Baker, and L. Bejder. 2011. Characteristics of a manta ray *Manta alfredi* population off Maui, Hawaii, and implications for management. Marine Ecology Progress Series 429:245-260.

Dean, R. G. 1991. Equilibrium beach profiles: Characteristics and applications. Journal of Coastal Research 7(1):53-84.

Deepwater Horizon Natural Resource Damage Assessment Trustees. 2016. Deepwater Horizon oil spill: Final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. U.S. Department of Commerce, National

Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

Dennison, W. C. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. Aquatic Botany 27(1):15-26.

DERM. 2012. Relocation of Acroporid corals from the Miami Beach-North pipeline corridor in association with the Miami-Dade test beach nourishment. Miami-Dade County Department of Environmental Resource Management, FDEP JCP Permit File #0295427-001JC, Miami, Florida.

Derraik, J. G. B. 2002. The pollution of the marine environment by plastic debris: A review. Marine Pollution Bulletin 44(9):842-852.

Dewar, H., and coauthors. 2008. Movements and site fidelity of the giant manta ray, *Manta birostris*, in the Komodo Marine Park, Indonesia. Marine Biology 155(2):121-133.

di Carlo, G., F. Badalamenti, A. C. Jensen, E. W. Koch, and S. Riggio. 2005. Colonisation process of vegetative fragments of *Posidonia oceanica* (L.) Delile on rubble mounds. Marine Biology 147(6):1261-1270.

Dickerson, C., D. G. Clarke, R. M. Engler, and K. J. Reine. 2001. Characterization of underwater sounds produced by bucket dredging operations. Engineer Research and Development Center, ERDC TN-DOER-E14, Vicksburg, MS.

Dickerson, D. D. 2013. Observed takes of sturgeon from dredging operations along the Atlantic and Gulf Coasts. U.S. Army Engineer Research and Development Center Environmental Laboratory, Vicksburg, MS.

Dickerson, D. D., K. J. Reine, D. A. Nelson, and C. E. Dickerson Jr. 1995. Assessment of sea-turtle abundance in six South Atlantic U.S. channels: Final report. U.S. Department of Defense, Army Corps of Engineers, Waterways Experiment Station, Miscellaneous Paper EL-95-5, Vicksburg, MS.

Dickerson, D. D., and coauthors. 2007. Effectiveness of relocation trawling during hopper dredging for reducing incidental take of sea turtless. U.S. Army Corps of Engineers, Engineer Research and Development Center Research Initiatives and Central Dredging Association, Lake Buena Vista, FL.

Dickerson, D. D., M. Wolters, and C. Theriot. 2008. Commitments of the Corps of Engineers: Navigation, dredging, and sea turtles. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southest Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-567, Miami, FL.

Diemer, K. M., B. Q. Mann, and N. E. Hussey. 2011. Distribution and movement of scalloped hammerhead *Sphryna lewini* and smooth hammerhead Sphyrna zygaena sharks along the east coast of southern Africa. African Journal of Marine Science 33(2):229-238.

DiJohnson, A. M. 2019. Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) behavioral responses to vessel traffic and habitat use in the Delaware River, USA. Master's Thesis. Delaware State University, Dover, DE.

Dodd Jr., C. K. 1988. Synopsis of the biological data on the loggerhead sea turtle: *Caretta caretta* (Linnaeus 1758). U.S. Department of the Interior, Fish and Wildlife Service, FAO Synopsis NMFS-149, Gainesville, FL.

Dodd, M. 2003. Summary of strandings during I-beam operation in Brunswick Ship Channel, Brunswick, GA during winter/spring 2003. Georgia Department of Natural Resources, Brunswick, Georgia

Dodge, R. E., and J. R. Vaisnys. 1977. Coral populations and growth patterns: Responses to sedimentation and turbidity associated with dredging. Journal of Marine Research 35(4):715-730.

Doney, S. C., and coauthors. 2012. Climate change impacts on marine ecosystems. Annual Review of Marine Science 4:11-37.

Doughty, R. W. 1984. Sea turtles in Texas: A forgotten commerce. The Southwestern Historical Quarterly 88(1):43-70.

Dovel, W., A. Pekovitch, and T. Berggren. 1992. Biology of the shortnose sturgeon (*Acipenser brevirostrum* Lesueur, 1818) in the Hudson River estuary, New York. Pages 187-216 *in* C. L. Smith, editor. Estuarine Research in the 1980s. State University of New York Press, Albany, New York.

Dovel, W. L., and T. J. Berggren. 1983. Atlantic sturgeon of the Hudson River estuary, New York. New York Fish and Game Journal 30:140-172.

Dow, W., K. L. Eckert, M. Palmer, and P. Kramer. 2007. An atlas of sea turtle nesting habitat for the wider Caribbean region. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy, WIDECAST Technical Report No. 6, Beaufort, NC.

Drevnick, P. E., and M. B. Sandheinrich. 2003. Effects of dietary methylmercury on reproductive endocrinology of fathead minnows. Environmental Science and Technology 37(19):4390-4396.

Dulvy, N. K., and coauthors. 2016. Ghosts of the coast: global extinction risk and conservation of sawfishes. Aquatic Conservation: Marine and Freshwater Ecosystems 26(1):134-153.

Dulvy, N. K., S. A. Pardo, C. A. Simpfendorfer, and J. K. Carlson. 2014. Diagnosing the dangerous demography of manta rays using life history theory. PeerJ Preprints 2.

Duncan, K. M., and K. N. Holland. 2006. Habitat use, growth rates and dispersal patterns of juvenile scalloped hammerhead sharks *Sphyrna lewini* in a nursery habitat. Marine Ecology Progress Series 312:211-221.

Duncan, M. S., J. J. Isely, and D. W. Cooke. 2004. Evaluation of shortnose sturgeon spawning in the Pinopolis Dam Tailrace, South Carolina. North American Journal of Fisheries Management 24:932–938.

Duncan, W. W., M. C. Freeman, C. A. Jennings, and J. T. McLean. 2003. Considerations for flow alternatives that sustain Savannah River fish populations. The University of Georgia, Institute of Ecology, Athens, GA.

Dunton, K. J., A. Jordaan, K. A. McKown, D. O. Conover, and M. G. Frisk. 2010. Abundance and distribution of Atlantic sturgeon (*Acipenser oxyrinchus*) within the Northwest Atlantic Ocean, determined from five fishery-independent surveys. Fishery Bulletin 108(4):450-465.

Duque, V. M., V. P. Páez, and J. A. Patino. 2000. Ecología de anidación y conservación de la tortuga caná, *Dermochelys coriacea*, en la Playona, Golfo de Urabá Chocoano (Colombia), en 1998 Actualidades Biologicas Medellín 22(72):37-53.

Durako, M. J., J. I. Kunzelman, W. J. Kenworthy, and K. K. Hammerstrom. 2003. Depth-related variability in the photobiology of two populations of *Halophila johnsonii* and *Halophila decipiens*. Marine Biology 142(6):1219-1228.

Dutton, D. L., P. H. Dutton, M. Chaloupka, and R. H. Boulon. 2005. Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biological Conservation 126(2):186-194.

Dwyer, K. L., C. E. Ryder, and R. Prescott. 2003. Anthropogenic mortality of leatherback turtles in Massachusetts waters. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Eckert, K. L., and S. A. Eckert. 1990. Embryo mortality and hatch success in (*in situ*) and translocated leatherback sea turtle (*Dermochelys coriacea*) eggs. Biological Conservation 53:37-46.

Eckert, K. L., S. A. Eckert, T. W. Adams, and A. D. Tucker. 1989a. Inter-nesting migrations by leatherback sea turtles (*Dermochelys coriacea*) in the West Indies. Herpetologica 45(2):190-194.

Eckert, K. L., B. P. Wallace, J. G. Frazier, S. A. Eckert, and P. C. H. Pritchard. 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of the Interior, Fish and Wildlife Service, BTP-R4015-2012, Washington, D.C.

Eckert, S. A. 2006. High-use oceanic areas for Atlantic leatherback sea turtles (*Dermochelys coriacea*) as identified using satellite telemetered location and dive information. Marine Biology 149(5):1257-1267.

Eckert, S. A., and coauthors. 2006. Internesting and postnesting movements and foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. Chelonian Conservation and Biology 5(2):239-248.

Eckert, S. A., K. L. Eckert, P. Ponganis, and G. L. Kooyman. 1989b. Diving and foraging behavior of leatherback sea turtles (*Dermochelys coriacea*). Canadian Journal of Zoology 67(11):2834-2840.

Eckert, S. A., D. W. Nellis, K. L. Eckert, and G. L. Kooyman. 1984. Deep diving record for leatherbacks. Marine Turtle Newsletter 31:4.

Eckert, S. A., and L. Sarti. 1997. Distant fisheries implicated in the loss of the world's largest leatherback nesting population. Marine Turtle Newsletter 78:2-7.

Edmunds, P. J., and R. Elahi. 2007. The demographics of a 15-year decline in cover of the Caribbean reef coral *Montastraea annularis*. Ecological Monographs 77(1):3-18.

Eguchi, T., P. H. Dutton, S. A. Garner, and J. Alexander-Garner. 2006. Estimating juvenile survival rates and age at first nesting of leatherback turtles at St. Croix, U.S. Virgin Islands. Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Ehrhart, L. M. 1983. Marine turtles of the Indian River Lagoon system. Florida Scientist 46(3/4):337-346.

Ehrhart, L. M., W. E. Redfoot, and D. A. Bagley. 2007. Marine turtles of the central region of the Indian River Lagoon system, Florida. Florida Scientist 70(4):415-434.

Ehrhart, L. M., and R. G. Yoder. 1978. Marine turtles of Merritt Island National Wildlife Refuge, Kennedy Space Centre, Florida. Florida Marine Research Publications 33:25-30.

Eiseman, N. J., and C. McMillan. 1980. A new species of seagrass, *Halophila johnsonii*, from the Atlantic coast of Florida. Aquatic Botany 9:15-19.

Engineer Research and Development Center. 2003. Bed-leveling following dredging operations. U.S. Department of Defense, U.S. Army Corps of Engineers, South Atlantic Division, Engineer Research and Development Center, Dredging Operations Technical Support (DOTS) Program, Request for Technical Assistance, Vicksburg, MS.

Epperly, S., and coauthors. 2002. Analysis of Sea Turtle Bycatch in the Commerical Shrimp Fisheries of the Southeast U.S. Waters and the Gulf of Mexico. NOAA Tech Memo NMFS-SEFSC-490. National Marine Fisheries Service, Miami, FL.

Epperly, S. P., J. Braun-McNeill, and P. M. Richards. 2007. Trends in catch rates of sea turtles in North Carolina, USA. Endangered Species Research 3:283-293.

Erbe, C., and C. McPherson. 2017. Underwater noise from geotechnical drilling and standard penetration testing. The Journal of the Acoustical Society of America 142(3):EL281-EL285.

ERC. 2012. Acoustic Telemetry Study of the Movements of Juvenile Sturgeons in Reach B of the Delaware River During Dredging Operations. Prepared for USACE, Philadelphia District. Contract No. W912BU-06-D-0003 Task Order No. 81.

ERC, I. 2006. Final report of shortnose sturgeon population studies in the Delaware River, January 1999 through March 2003. Prepared for NOAA Fisheries and NJ Division of Fish and Wildlife. 11 pp.

Erftemeijer, P. L. A., B. Riegl, B. W. Hoeksema, and P. A. Todd. 2012. Environmental impacts of dredging and other sediment disturbances on corals: A review. Marine Pollution Bulletin 64(9):1737-1765.

Erickson, D. L., and coauthors. 2011. Use of pop-up satellite archival tags to identify oceanic-migratory patterns for adult Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus* Mitchell, 1815. Journal of Applied Ichthyology 27(2):356-365.

Evans, P. G. H., and A. Bjørge. 2013. Impacts of climate change on marine mammals. Marine Climate Change Impacts Partnership Science Review:134-148.

Evermann, B. W., and B. A. Bean. 1897. Indian River and its fishes. Pages 227-248 *in* Report of the Commissioner of Fish and Fisheries for the Fiscal Year Ending June 30, 1896, volume 22. U.S. Commission of Fish and Fisheries and NOAA Central Library Data Imaging Project, Washington, D.C.

Evermann, B. W., and B. A. Bean. 1898. Indian River and its fishes. U.S. Commission on Fish and Fisheries 22:227-248.

Fairbanks, R. G. 1989. A 17,000-year glacio-eustatic sea level record: Influence of glacial melting rates on the Younger Dryas event and deep-ocean circulation. Nature 342(6250):637-642.

Farrae, D. J., W. C. Post, and T. L. Darden. 2017a. Genetic characterization of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, in the Edisto River, South Carolina and identification of genetically discrete fall and spring spawning. Conservation Genetics:1-11.

Farrae, D. J., W. C. Post, and T. L. Darden. 2017b. Genetic characterization of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus,* in the Edisto River, South Carolina and identification of genetically discrete fall and spring spawning. Conservation Genetics 18(4):813-823.

Feldheim, K. A., A. T. Fields, D. D. Chapman, R. M. Scharer, and G. R. Poulakis. 2017. Insights into reproduction and behavior of the smalltooth sawfish Pristis pectinata. Endangered Species Research 34:463-471.

Fernandes, S. J., G. B. Zydlewski, J. D. Zydlewski, G. S. Wippelhauser, and M. T. Kinnison. 2010. Seasonal distribution and movements of shortnose sturgeon and Atlantic sturgeon in the Penobscot River Estuary, Maine. Transactions of the American Fisheries Society 139(5):1436-1449.

Ferraroli, S., J.-Y. Georges, P. Gaspar, and Y. Le Maho. 2004. Where leatherback turtles meet fisheries. Nature 429:521-522.

Fish, M. R., and coauthors. 2005. Predicting the impact of sea-level rise on Caribbean sea turtle nesting habitat. Conservation Biology 19(2):482-491.

Fisher, M. T. 2009. Atlantic sturgeon progress report. Delaware Division of Fish and Wildlife, Department of Natural Resources and Environmental Control, Delaware State Wildlife Grant, Project T-4-1, December 16, 2008 to December 15, 2009, Dover, DE.

Fisher, M. T. 2011. Atlantic sturgeon progress report. Delaware Division of Fish and Wildlife, Department of Natural Resources and Environmental Control, Delaware State Wildlife Grant, Project T-4-1, October 1, 2006 to October 15, 2010, Dover, DE.

FitzSimmons, N. N., L. W. Farrington, M. J. McCann, C. J. Limpus, and C. Moritz. 2006. Green turtle populations in the Indo-Pacific: A (genetic) view from microsatellites. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-536, Miami, FL.

Fleming, J. E., T. D. Bryce, and J. P. Kirk. 2003. Age, growth and status of shortnose sturgeon in the lower Ogeechee River, Georgia. Fish and Wildlife Agencies

Flournoy, P. H., S. G. Rogers, and P. S. Crawford. 1992a. Restoration of shortnose sturgeon in the Altamaha River, Georgia. U.S. Fish and Wildlife Service, Atlanta, Georgia.

Flournoy, P. H., S. G. Rogers, and P. S. Crawford. 1992b. Restoration of shortnose sturgeon in the Altamaha River, Georgia: Final report. U.S. Fish and Wildlife Service, Atlanta, GA.

Fogarty, N. D., S. V. Vollmer, and D. R. Levitan. 2012. Weak prezygotic isolating mechanisms in threatened Caribbean *Acropora* corals. PLOS ONE 7(2):e30486.

Foley, A. M., B. A. Schroeder, and S. L. MacPherson. 2008. Post-nesting migrations and resident areas of Florida loggerheads (*Caretta caretta*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-582, Miami, FL.

Foley, A. M., B. A. Schroeder, A. E. Redlow, K. J. Fick-Child, and W. G. Teas. 2005. Fibropapillomatosis in stranded green turtles (*Chelonia mydas*) from the eastern United States (1980-98): Trends and associations with environmental factors. Journal of Wildlife Diseases 41(1):29-41.

Foley, A. M., K. E. Singel, P. H. Dutton, T. M. Summers, and A. E. Redlow. 2007. Characteristics of a green turtle (*Chelonia mydas*) assemblage in northwestern Florida determined during a hypothermic stunning event. Gulf of Mexico Science 25(2):13.

Fong, P., and D. Lirman. 1995. Hurricanes cause population expansion of the branching coral *Acropora palmata* (Scleractinia): Wound healing and growth patterns of asexual recruits. Marine Ecology 16(4):317-335.

Fonseca, M. S., W. J. Kenworthy, and P. E. Whitfield. 2000. Temporal dynamics of seagrass landscapes: A preliminary comparison of chronic and extreme disturbance events, volume 7. Biologia Marina Mediterranea, Instituto di Zoologia, Genova, Italy.

Formia, A. 1999. Les tortues marines de la Baie de Corisco. Canopee 14: i-ii.

Fourney, F., and J. Figueiredo. 2017. Additive negative effects of anthropogenic sedimentation and warming on the survival of coral recruits. Scientific Reports 7(1):12380.

Fox, A. G., D. Peterson, I. Wirgin, and A. Cummins. 2019. Quantifying Annual Recruitment and Nursery Habitats of Atlantic Sturgeon in Georgia. National Marine Fisheries Service.

Fox, A. G., and D. L. Peterson. 2017. Occurrence and Movements of Atlantic and Shortnose Sturgeon in Cumberland Sound and the St. Marys River, Georgia. University of Georgia.

Fox, A. G., E. S. Stowe, and D. L. Peterson. 2017. Occurrence and Movements of Shortnose and Atlantic Sturgeon in the St. Johns River, Florida. Warnell School of Forestry and Natural Resources, University of Georgia, Athens, GA.

Frankham, R., C. J. A. Bradshaw, and B. W. Brook. 2014. Genetics in conservation management: revised recommendations for the 50/500 rules, Red List criteria and population viability analyses. Biological Conservation 170:56-63.

Frazer, N. B., and L. M. Ehrhart. 1985. Preliminary growth models for green, (*Chelonia mydas*), and loggerhead, (*Caretta caretta*), turtles in the wild. Copeia 1985(1):73-79.

Fretey, J. 2001. Biogeography and conservation of marine turtles of the Atlantic coast of Africa. Convention on the Conservation of Migratory Species of Wild Animals (CMS), Secretariat, Bonn, Germany.

Fretey, J., A. Billes, and M. Tiwari. 2007. Leatherback, *Dermochelys coriacea*, nesting along the Atlantic coast of Africa. Chelonian Conservation and Biology 6(1):126-129.

Fritts, M., and D. Peterson. 2010. Status of Atlantic sturgeon and shortnose sturgeon in the St. Marys and Satilla Rivers, Georgia. Final report ot the National Marine Fisheries Service. Warnell School of Forestry and Natural Resources - University of Georgia.

Fritts, M. W., C. Grunwald, I. Wirgin, T. L. King, and D. L. Peterson. 2016. Status and genetic character of Atlantic sturgeon in the Satilla River, Georgia. Transactions of the American Fisheries Society 145(1):69-82.

Fritts, T. H., and M. A. McGehee. 1982. Effects of petroleum on the development and survival of marine turtles embryos. U.S. Department of the Interior, U.S. Fish and Wildlife Service, FWS/OBS-82/37, Contract No. 14-16-0009-80-946, Belle Chasse, LA.

Fujihara, J., T. Kunito, R. Kubota, and S. Tanabe. 2003. Arsenic accumulation in livers of pinnipeds, seabirds and sea turtles: Subcellular distribution and interaction between arsenobetaine and glycine betaine. Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 136(4):287-296.

Gall, S. C., and R. C. Thompson. 2015. The impact of debris on marine life. Mar Pollut Bull 92(1-2):170-179.

Gallegos, C. L., and W. J. Kenworthy. 1996. Seagrass depth limits in the Indian River Lagoon (Florida, USA): Application of the optical water quality model. Estuarine, Coastal, and Shelf Science 42(3):267-288.

Gallo, F., and coauthors. 2018. Marine litter plastics and microplastics and their toxic chemicals components: The need for urgent preventive measures. Environmental Sciences Europe 30(1):13.

García-Fernández, A. J., and coauthors. 2009. Heavy metals in tissues from loggerhead turtles (*Caretta caretta*) from the southwestern Mediterranean (Spain). Ecotoxicology and Environmental Safety 72(2):557-563.

García-Muñoz, D., and L. Sarti. 2000. Reproductive cycles of leatherback turtles. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-436, Miami, FL.

Garcia-Sais, J., and coauthors. 2008. The state of coral reef ecosystems of Puerto Rico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team, NOAA Technical Memorandum NOS NCCOS 73, Silver Spring, MD.

Garcia Reyes, J., and N. V. Schizas. 2010. No two reefs are created equal: Fine-scale population structure in the threatened coral species *Acropora palmata* and *A. cervicornis*. Aquatic Biology 10(1):69-83.

García Sais, J. R., S. Williams, R. Esteves, J. Sabater Clavell, and M. Carlo. 2013. Synoptic survey of Acroporid corals in Puerto Rico, 2011-2013: Final Report. Puerto Rico Department of Natural and Environmental Resources.

Gardner, S. C., S. L. Fitzgerald, B. A. Vargas, and L. M. Rodríguez. 2006. Heavy metal accumulation in four species of sea turtles from the Baja California Peninsula, Mexico. BioMetals 19(1):91-99.

Garrett, C. L. 2004. Priority substances of interest in the Georgia Basin - Profiles and background information on current toxics issues. Canadian Toxics Work Group Puget Sound, Georgia Basin International Task Force, GBAP Publication No. EC/GB/04/79.

Geraci, J. R. 1990. Physiologic and toxic effects on cetaceans. Pages 167-197 *in* J. R. Geraci, and D. J. St. Aubin, editors. Sea Mammals and Oil: Confronting the Risks. Academic Press, San Diego, CA.

Germanov, E. S., and A. D. Marshall. 2014. Running the gauntlet: regional movement patterns of Manta alfredi through a complex of parks and fisheries. PLOS ONE 9(10):e110071.

Germanov, E. S., and coauthors. 2019. Microplastics on the menu: Plastics pollute Indonesian manta ray and whale shark feeding grounds. Frontiers in Marine Science 6(679).

Giesy, J. P., J. Newsted, and D. L. Garling. 1986. Relationships between chlorinated hydrocarbon concentrations and rearing mortality of Chinook salmon (*Oncorhynchus tshawytscha*) eggs from Lake Michigan. Journal of Great Lakes Research 12(1):82-98.

Gilbert, C. R. 1989a. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic Bight). Atlantic and shortnose sturgeons. University of Florida, Florida Museum of Natural History, USFWS Biological Report 82(11.122) and USACE TR EL-82-4, Gainesville, FL.

Gilbert, C. R. 1989b. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic Bight): Atlantic and shortnose sturgeons. U.S. Department of the Interior, Fish and Wildlife Service and U.S. Army Corps of Engineers, Waterways Experiment Station, Washington, D. C.

Gilman, E. L., J. Ellison, N. C. Duke, and C. Field. 2008. Threats to mangroves from climate change and adaptation options: A review. Aquatic Botany 89(2):237-250.

Gilmore, G. R. 1995. Environmental and Biogeographic Factors Influencing Ichthyofaunal Diversity: Indian River Lagoon. Bulletin of Marine Science 57(1):153-170.

Gilmore, M. D., and B. R. Hall. 1976. Life history, growth habits, and constructional roles of *Acropora cervicornis* in the patch reef environment. Journal of Sedimentary Petrology 46(3):519-522.

Gilmour, J. 1999. Experimental investigation into the effects of suspended sediment on fertilisation, larval survival and settlement in a scleractinian coral. Marine Biology 135(3):451-462.

Ginger. 2012. Suivi de l'incidence de la technique de remobilisation des sédiments par injection d'eau. Pour le Grand Port Maritime de Bordeaux.

Girard, C., A. D. Tucker, and B. Calmettes. 2009. Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: Dispersal in highly dynamic conditions. Marine Biology 156(9):1827-1839.

Girondot, M., and coauthors. 2015. Spatio-temporal distribution of *Manta birostris* in French Guiana waters. Journal of the Marine Biological Association of the United Kingdom 95(1):153-160.

Godley, B. J., D. R. Thompson, and R. W. Furness. 1999. Do heavy metal concentrations pose a threat to marine turtles from the Mediterranean Sea? Marine Pollution Bulletin 38(6):497-502.

Goff, G. P., and J. Lien. 1988. Atlantic leatherback turtles, *Dermochelys coriacea*, in cold water off Newfoundland and Labrador. Canadian Field-Naturalist 102:1-5.

Goldberg, W. M. 1973. The ecology of the coral octocoral communities off the southeast Florida coast: Geomorphology, species composition, and zonation. Bulletin of Marine Science 23(3):465-488.

Gomez, C., and coauthors. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: The disparity between science and policy. Canadian Journal of Zoology 94(12):801-819.

González-Díaz, P., G. González-Sansón, S. Álvarez Fernández, and O. Perera Pérez. 2010. High spatial variability of coral, sponges and gorgonian assemblages in a well preserved reef. Revista de biología tropical 58(2):621-634.

Gonzalez Carman, V., and coauthors. 2011. Argentinian coastal waters: A temperate habitat for three species of threatened sea turtles. Marine Biology Research 7:500-508.

Gordon, A. N., A. R. Pople, and J. Ng. 1998. Trace metal concentrations in livers and kidneys of sea turtles from south-eastern Queensland, Australia. Marine and Freshwater Research 49(5):409-414.

Goreau, T. F. 1959. The ecology of Jamaican coral reefs I. Species composition and zonation. Ecology 40(1):67-90.

Gowan, T. A., and J. G. Ortega-Ortiz. 2014. Wintering habitat model for the North Atlantic right whale (*Eubalaena glacialis*) in the southeastern United States. PLOS ONE 9(4):e95126.

Graham, J. E., and R. van Woesik. 2013. The effects of partial mortality on the fecundity of three common Caribbean corals. Marine Biology 160:2561-2565.

Graham, N. A. J., and coauthors. 2008. Climate Warming, Marine Protected Areas and the Ocean-Scale Integrity of Coral Reef Ecosystems. PLOS ONE 3(8):e3039.

Graham, R. T., and coauthors. 2012. Satellite tracking of manta rays highlights challenges to their conservation. PLOS ONE 7(5).

Graham, T. R. 2009. Scyphozoan jellies as prey for leatherback sea turtles off central California. Master's Thesis. San José State University, Moss Landing, CA.

Grant, S. C. H., and P. S. Ross. 2002. Southern resident killer whales at risk: Toxic chemicals in the British Columbia and Washington environment. Fisheries and Oceans Canada, Institute of Ocean Sciences, Canadian Technical Report of Fisheries and Aquatic Sciences 2412, Sidney, British Columbia, Canada.

Green, D. 1993. Growth rates of wild immature green turtles in the Galápagos Islands, Ecuador. Journal of Herpetology 27(3):338-341.

Greene, K. E., J. L. Zimmerman, R. W. Laney, and J. C. Thomas-Blate. 2009. Atlantic coast diadromous fish habitat: A review of utilization, threats, recommendations for conservation, and research needs. Atlantic States Marine Fisheries Commission Washington, D.C.

Greenlee, B., and D. H. Secor. 2016. Assessment of critical habitats for recovering the Chesapeake Bay Atlantic sturgeon Distinct Population Segment: Final report. Virginia Department of Game and Inland Fisheries, Grant # NA13NMF4720037, Charles City, VA.

Greer Jr., A. E., J. D. Lazell Jr., and R. M. Wright. 1973. Anatomical evidence for a counter-current heat exchanger in the leatherback turtle (*Dermochelys coriacea*). Nature 244:181.

Grober-Dunsmore, R., V. Bonito, and T. K. Frazer. 2006. Potential inhibitors to recovery of *Acropora palmata* populations in St. John, US Virgin Islands. Marine Ecology Progress Series 321:123-132.

Groombridge, B., and L. Wright (compilers). 1982. Kemp's ridley or Atlantic ridley, *Lepidochelys kempii* (Garman 1880). Pages 201-208 *in* The IUCN Amphibia-Reptilia Red Data Book. Part 1, Testudines, Crocodylia, Rhynchocephalia, volume 1. IUCN Conservation Monitoring Centre, Gland, Switzerland.

Grunwald, C., L. Maceda, J. Waldman, J. Stabile, and I. Wirgin. 2008. Conservation of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*: Delineation of stock structure and distinct population segments. Conservation Genetics 9(5):1111-1124.

Gudger, E. W. 1922. The most northerly record of the capture in Atlantic waters of the United States of the giant ray, *Manta birostris*. Science 55(1422):338-340.

Guilbard, F., J. Munro, P. Dumont, D. Hatin, and R. Fortin. 2007. Feeding ecology of Atlantic sturgeon and lake sturgeon co-occurring in the St. Lawrence estuarine transition zone. American Fisheries Society Symposium 56:85-104.

Guinder, V. A., and J. C. Molinero. 2013. Climate change effects on marine phytoplankton. Pages 68-90 *in* A. H. Arias, and M. C. Menendez, editors. Marine Ecology in a Changing World. CRC Press, Boca Raton, FL.

Guseman, J. L., and L. M. Ehrhart. 1991. Ecological geography of western Atlantic loggerheads and green turtles: Evidence from remote tag recoveries. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Technical Memorandum NMFS-SEFSC-302, Miami, FL.

Guzmán, H. M. 1991. Restoration of coral reefs in Pacific Costa Rica. Conservation Biology 5(2):189-194.

GWC. 2006a. Georgia Water Coalition. Interbasin Transfer Fact Sheet. http://www.garivers.org/gawater/pdf%20files/IBT%20fact%20sheet02-06.pdf.

GWC. 2006b. Interbasin Transfer Fact Sheet. Georgia Water Coalition, http://www.garivers.org/gawater/pdf%20files/IBT%20fact%20sheet02-06.pdf.

Hager, C., J. Kahn, C. Watterson, J. Russo, and K. Hartman. 2014. Evidence of Atlantic sturgeon spawning in the York River system. Transactions of the American Fisheries Society 143(5):1217-1219.

Hale, E. A., and coauthors. 2016. Abundance estimate for and habitat use by early juvenile Atlantic sturgeon within the Delaware River Estuary. Transactions of the American Fisheries Society 145(6):1193-1201.

Hall, J. W., T. I. J. Smith, and S. D. Lamprecht. 1991. Movements and habitats of shortnose sturgeon, *Acipenser brevirostrum* in the Savannah River. Copeia (3):695-702.

Hall, L. M., M. D. Hanisak, and R. W. Virnstein. 2006. Fragments of the seagrasses *Halodule wrightii* and *Halophila johnsonii* as potential recruits in Indian River Lagoon, Florida. Marine Ecology Progress Series 310:109-117.

Halvorsen, M. B., B. M. Casper, F. Matthews, T. J. Carlson, and A. N. Popper. 2012a. Effects of exposure to pile-driving sounds on the lake sturgeon, Nile tilapia and hogchoker. Proceedings of the Royal Society B: Biological Sciences 279(1748):4705-4714.

Halvorsen, M. B., B. M. Casper, C. M. Woodley, T. J. Carlson, and A. N. Popper. 2012b. Threshold for onset of injury in Chinook salmon from exposure to impulsive pile driving sounds. PLOS ONE 7(6):e38968.

Hammerschmidt, C. R., M. B. Sandheinrich, J. G. Wiener, and R. G. Rada. 2002. Effects of dietary methylmercury on reproduction of fathead minnows. Environmental Science and Technology 36(5):877-883.

Hammerstom, K. K., and W. J. Kenworthy. 2003. Investigating the existence of a *Halophila johnsonii* sediment seed bank. Johnson’s Seagrass Implementation Team.

Hansen, D. J. 1985. Environmental assessment of the effects of offshore oil development on marine mammals occurring in Alaska marine waters, Vancouver, British Columbia, Canada.

Hare, J. A., and coauthors. 2016. A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. Continental Shelf. PLOS ONE 11(2):30.

Harms, C. A., K. M. Mallo, P. M. Ross, and A. Segars. 2003. Venous blood gases and lactates of wild loggerhead sea turtles (*Caretta caretta*) following two capture techniques. Journal of Wildlife Diseases 39(2):366-374.

Hart, K. M., M. M. Lamont, I. Fujisaki, A. D. Tucker, and R. R. Carthy. 2012. Common coastal foraging areas for loggerheads in the Gulf of Mexico: Opportunities for marine conservation. Biological Conservation 145(1):185-194.

Hartwell, S. I. 2004. Distribution of DDT in sediments off the central California coast. Marine Pollution Bulletin 49(4):299-305.

Hastings, M. C., and A. N. Popper. 2005. Effects of sound on fish. Jones & Stokes, California Department of Transportation Contract No. 43A0139, Task Order 1, Sacramento, CA.

Hatcher, B. G. 1997. Coral reef ecosystems: How much greater is the whole than the sum of the parts? Coral Reefs 16(S1):S77-S91.

Hatin, D., J. Munro, F. Caron, and R. D. Simons. 2007. Movements, home range size, and habitat use and selection of early juvenile Atlantic sturgeon in the St. Lawrence estuarine transition zone. American Fisheries Society Symposium 56:129-155.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology 13(5):923-932.

Hayhoe, K., and coauthors. 2017. Climate models, scenarios, and projections. Pages 133-160 *in* D. J. Wuebbles, and coeditors, editors. Climate Science Special Report: Fourth National Climate Assessment, volume I. U.S. Global Change Research Program, Washington, D.C.

Hays, G. C., and coauthors. 2001. The diving behavior of green turtles undertaking oceanic migration to and from Ascension Island: Dive durations, dive profiles, and depth distribution. Journal of Experimental Biology 204:4093-4098.

Hays, G. C., and coauthors. 2002. Water temperature and internesting intervals for loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtles. Journal of Thermal Biology 27(5):429-432.

Hays, G. C., J. D. R. Houghton, and A. E. Myers. 2004. Pan-Atlantic leatherback turtle movements. Nature 429:522.

Hearn, A. R., and coauthors. 2014. Elasmobranchs of the Galapagos Marine Reserve. Pages 23-59 *in* J. Denkinger, and L. Vinueza, editors. Social and Ecological Interactions in the Galapagos Island, The Galapagos Marine Reserve: A dynamic social-ecological system. Springer, New York, NY.

Hearn, A. R., J. T. Ketchum, A. P. Klimley, E. Espinoza, and C. Peñaherrera. 2010. Hotspots within hotspots? Hammerhead shark movements around Wolf Island, Galapagos Marine Reserve. Marine Biology 157(9):1899-1915.

Heidelbaugh, W. S., and coauthors. 2000. Reciprocal transplanting of the threatened seagrass *Halophila johnsonii* (Johnson's Seagrass) in the Indian River Lagoon, Florida. Pages 197-210 *in* S. A. Bortone, editor. Seagrasses: Monitoring, Ecology, Physiology, and Management. CRC Press, Boca Raton, FL.

Heinrichs, S., M. O'Malley, H. Medd, and P. Hilton. 2011. Global Threat to Manta and Mobula Rays. Manta Ray of Hope, 2011 Report.

Henshall, J. A., and United States Fish Commission. 1891. Report Upon a Collection of Fishes Made in Southern Florida during 1889. U.S. Government Printing Office, Washington, D.C.

Heppell, S. S., and coauthors. 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chelonian Conservation and Biology 4(4):767-773.

Heppell, S. S., L. B. Crowder, D. T. Crouse, S. P. Epperly, and N. B. Frazer. 2003a. Population models for Atlantic loggerheads: Past, present, and future. Pages 255-273 *in* A. B. Bolten, and B. E. Witherington, editors. Loggerhead Sea Turtles. Smithsonian Books, Washington, D.C.

Heppell, S. S., L. B. Crowder, and T. R. Menzel. 1999. Life table analysis of long-lived marine species with implications for conservation and management, volume 23. American Fisheries Society, Monterey, CA

Heppell, S. S., L. B. Crowder, and J. Priddy. 1995. Evaluation of a fisheries model for hawksbill sea turtle (*Eretmochelys imbricata*) harvest in Cuba. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-OPR-5, Raleigh, NC.

Heppell, S. S., M. L. Snover, and L. B. Crowder. 2003b. Sea turtle population ecology. Pages 275-306 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, Vol. II. CRC Press, Boca Raton, FL.

Herbst, L. H. 1994. Fibropapillomatosis of marine turtles. Annual Review of Fish Diseases 4:389-425.

Herbst, L. H. 1995. An infectious etiology for green turtle fibropapillomatosis, volume 36. American Association for Cancer Research, Toronto, Ontario, Canada.

Hernández-Delgado, E. A., and coauthors. 2011. Sediment stress, water turbidity, and sewage impacts on threatened elkhorn coral (*Acropora palmata*) stands at Vega Baja, Puerto Rico. Gulf and Caribbean Fisheries Institute, San Juan, Puerto Rico.

Heron, S. F., C. M. Eakin, J. A. Maynard, and R. van Hooidonk. 2016. Impacts and effects of ocean warming on coral reefs. Pages 177-197 *in* D. Laffoley, and J. M. Baxter, editors. Explaining Ocean Warming: Causes, scale, effects and consequences. IUCN, Gland, Switzerland.

Heron, S. F., J. Morgan, C. M. Eakin, and W. Skirving. 2008. Hurricanes and their effects on coral reefs. Pages 31-36 *in* C. Wilkinson, and D. Souter, editors. Status of Caribbean Coral Reefs after Bleaching and Hurricanes in 2005. Global Coral Reef Monitoring Network, Reef and Rainforest Research Center, Townsville, Australia.

Highsmith, R. C. 1982. Reproduction by fragmentation in corals. Marine Ecology Progress Series 7(2):207-226.

Hightower, J. E. 1998. Prioritizing habitat restoration efforts for anadromous fishes in North Carolina. U.S. Department of the Interior, U.S. Geological Survey, Biological Resources Division, Report to the NC Cooperative Fish and Wildlife Research Unit.

Hildebrand, H. H. 1963. Hallazgo del area de anidacion de la tortuga marina "lora", *Lepidochelys kempi* (Garman), en la costa occidental del Golfo de Mexico (Rept., Chel.). Ciencia, Mexico 22(4):105-112.

Hildebrand, H. H. 1982. A historical review of the status of sea turtle populations in the western Gulf of Mexico. Pages 447-453 *in* K. A. Bjorndal, editor. Biology and Conservation of Sea Turtles: Proceedings of the World Conference on Sea Turtle Conservation. Smithsonian Institution Press, Washington, D.C.

Hildebrand, J. A. 2009. Metrics for characterizing the sources of ocean anthropogenic noise. Journal of the Acoustical Society of America 125(4).

Hilterman, M., E. Goverse, M. Godfrey, M. Girondot, and C. Sakimin. 2003. Seasonal sand temperature profiles of four major leatherback nesting beaches in the Guyana Shield. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Hirth, H. F. 1971. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus) 1758. Food and Agricultural Organization of the United Nations, FAO Fisheries Synopsis No. 85, Rome, Italy.

Hirth, H. F. 1997. Synopsis of biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). University of Utah, Salt Lake City, UT.

Hirth, H. F., J. Kasu, and T. Mala. 1993. Observations on a leatherback turtle *Dermochelys coriacea* nesting population near Piguwa, Papua New Guinea. Biological Conservation 65(1):77-82.

Hollensead, L. D., R. D. Grubbs, J. K. Carlson, and D. M. Bethea. 2016. Analysis of fine-scale daily movement patterns of juvenile Pristis pectinata within a nursery habitat. Aquatic Conservation: Marine and Freshwater Ecosystems 26(3):492-505.

Hollensead, L. D., R. D. Grubbs, J. K. Carlson, and D. M. Bethea. 2018. Assessing residency time and habitat use of juvenile smalltooth sawfish using acoustic monitoring in a nursery habitat. Endangered Species Research 37:119-131.

Holt, M. M. 2008. Sound exposure and southern resident killer whales (*Orcinus orca*): A review of current knowledge and data gaps. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, NOAA Technical Memorandum NMFS-NWFSC-89, Seattle, WA.

Holton Jr., J. W., and J. B. Walsh. 1995. Long-term dredged material management plan for the upper James River, Virginia. Waterway Surveys and Engineering, Ltd, Virginia Beach, VA.

Hoover, J. J., K. A. Boysen, J. A. Beard, and H. Smith. 2011. Assessing the risk of entrainment by cutterhead dredges to juvenile lake sturgeon (*Acipenser fulvescens*) and juvenile pallid sturgeon (*Scaphirhynchus albus*). Journal of Applied Ichthyology 27(2):369-375.

Houghton, J. D. R., T. K. Doyle, M. W. Wilson, J. Davenport, and G. C. Hays. 2006. Jellyfish aggregations and leatherback turtle foraging patterns in a temperate coastal environment. Ecology 87(8):1967-1972.

Hudson, J. H., and R. Diaz. 1988. Damage survey and restoration of M/V *Wellwood* grouding site, Molasses Reef, Key Largo National Marine Sanctuary, Florida, volume 2. 6th International Coral Reef Symposium Executive Committee, Townsville, Queensland, Australia.

Hughes, G. R. 1996. Nesting of the leatherback turtle (*Dermochelys coriacea*) in Tongaland, KwaZulu-Natal, South Africa, 1963-1995. Chelonian Conservation and Biology 2(2):153-158.

Hulme, P. E. 2005. Adapting to climate change: Is there scope for ecological management in the face of a global threat? Journal of Applied Ecology 42(5):784-794.

Huntington, B. E., M. Karnauskas, and D. Lirman. 2011. Corals fail to recover at a Caribbean marine reserve despite ten years of reserve designation. Coral Reefs 30(4):1077-1085.

Hussey, Gay, and Bell. 1975. Engineering report on agitation dredging in Savannah Harbor. State of Georgia, Savannah, GA.

Idjadi, J. A., and coauthors. 2006. Rapid phase-shift reversal on a Jamaican coral reef. Coral Reefs 25(2):209-211.

Ingle, R. M. 1952. Studies on the effect of dredging operations upon fish and shellfish. Florida Board of Conservation, Technical Series No. 5 (1-25).

Ingle, R. M., and F. G. W. Smith. 1974. Sea Turtles: and the Turtle Industry of the West Indies, Florida and the Gulf of Mexico, Revised Edition. University of Miami Press, Coral Gables, FL.

Ingram, E. C., and D. L. Peterson. 2016. Annual spawning migrations of adult Atlantic sturgeon in the Altamaha River, Georgia. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 8(1):595-606.

Intergovernmental Panel on Climate Change. 2007. Climate Change 2007: Synthesis report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Intergovernmental Panel on Climate Change. 2013. Climate Change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change, Cambridge, United Kingdom; New York, NY.

Intergovernmental Panel on Climate Change. 2014. Summary for policymakers, Cambridge, United Kingdom and New York, NY.

ISED. 2014. International Sawfish Encounter Database. Florida Museum of Natural History, Gainesville, Florida. http://www.flmnh.ufl.edu/fish/sharks/sawfish/sawfishdatabase.html.

Iwata, H., S. Tanabe, N. Sakai, and R. Tatsukawa. 1993. Distribution of persistent organochlorines in the oceanic air and surface seawater and the role of ocean on their global transport and fate. Environmental Science & Technology 27(6):1080-1098.

Jaap, W. C. 1984. The ecology of south Florida coral reefs: A community profile. U.S. Department of the Interior, Fish and Wildlife Service, Minerals Management Service, FWS/OBS-82/08 and MMS 84-0038, Washington, D.C.

Jaap, W. C., W. G. Lyons, P. Dustan, and J. C. Halas. 1989. Stony coral (Scleractinia and Milleporina) community structure at Bird Key Reef, Ft. Jefferson National Monument, Dry Tortugas, Florida. Florida Marine Research Institute, Number 46, Saint Petersburg, FL.

Jackson, J. B. C., M. K. Donovan, K. L. Cramer, and V. V. Lam (editors). 2014. Status and trends of Caribbean coral reefs: 1970-2012. Global Coral Reef Monitoring Network, IUCN, Gland, Switzerland.

Jackson, J. B. C., and coauthors. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293(5530):629-638.

Jacobson, E. R. 1990. An update on green turtle fibropapilloma. Marine Turtle Newsletter 49:7-8.

Jacobson, E. R., and coauthors. 1989. Cutaneous fibropapillomas of green turtles (*Chelonia mydas*). Journal of Comparative Pathology 101(1):39-52.

Jacobson, E. R., S. B. Simpson Jr, and J. P. Sundberg. 1991. Fibropapillomas in green turtles. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOAA Technical Memorandum NOAA-TM-NMFS-SWFSC-156, Honolulu, HI.

Jambeck, J. R., and coauthors. 2015. Plastic waste inputs from land into the ocean. Science 347(6223):768-771.

James, M. C., S. A. Eckert, and R. A. Myers. 2005. Migratory and reproductive movements of male leatherback turtles (*Dermochelys coriacea*). Marine Biology 147(4):845-853.

James, M. C., S. A. Sherrill-Mix, and R. A. Myers. 2007. Population characteristics and seasonal migrations of leatherback sea turtles at high latitudes. Marine Ecology Progress Series 337:245-254.

Jarvis, P. L., J. S. Ballantyne, and W. E. Hogans. 2001. The influence of salinity on the growth of juvenile shortnose sturgeon. North American Journal of Aquaculture 63(4):272 - 276.

Jenkins Jr., W. E., T. I. J. Smith, L. D. Heyward, and D. M. Knott. 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum*, juveniles to different salinity and dissolved oxygen concentrations, volume 47. Southeastern Association of Fish and Wildlife Agencies.

Jenkins, W. E., T. I. J. Smith, L. D. Heyward, and D. M. Knott. 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum*, juveniles to different salinity and dissolved oxygen concentrations. Pages 476-484 *in* Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Jessop, T. S., R. Knapp, J. M. Whittier, and C. J. Limpus. 2002. Dynamic endocrine responses to stress: Evidence for energetic constraints and status dependence in breeding male green turtles. General and Comparative Endocrinology 126(1):59-67.

Jewett-Smith, J., C. McMillan, W. J. Kenworthy, and K. Bird. 1997. Flowering and genetic banding patterns of *Halophila johnsonii* and conspecifics. Aquatic Botany 59(3-4):323-331.

Johnson, S. A., and L. M. Ehrhart. 1994. Nest-site fidelity of the Florida green turtle. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-341, Miami, FL.

Johnson, S. A., and L. M. Ehrhart. 1996. Reproductive ecology of the Florida green turtle: Clutch frequency. Journal of Herpetology 30(3):407-410.

Jones, G. A., and S. G. Candy. 1981. Effects of dredging on the macrobenthic infauna of Botany Bay. Australian Journal of Marine and Freshwater Research 32(3):379-398.

Jones, G. P., M. I. McCormick, M. Srinivasan, and J. V. Eagle. 2004. Coral decline threatens fish biodiversity in marine reserves. Proc Natl Acad Sci U S A 101(21):8251-8253.

Jones, T. T., M. D. Hastings, B. L. Bostrom, D. Pauly, and D. R. Jones. 2011. Growth of captive leatherback turtles, *Dermochelys coriacea*, with inferences on growth in the wild: Implications for population decline and recovery. Journal of Experimental Marine Biology and Ecology 399(1):84-92.

Jørgensen, E. H., Ø. Aas-Hansen, A. G. Maule, J. E. T. Strand, and M. M. Vijayan. 2004. PCB impairs smoltification and seawater performance in anadromous Arctic charr (*Salvelinus alpinus*). Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 138(2):203-212.

Júnior, T. V., C. M. Vooren, and R. P. Lessa. 2009. Feeding strategy of the night shark (*Carcharhinus signatus*) and scalloped hammerhead shark (*Sphyrna lewini*) near seamounts off northeastern Brazil. Brazilian Journal of Oceanography 57(2):97-104.

Kahn, A. E. 2019. Summary report for the southern Indian River Lagoon. Florida Fish and Wildlife Research Institute, Technical Report 17, Version 3, St. Petersburg, FL.

Kahn, A. E., J. L. Beal, and M. J. Durako. 2013. Diurnal and Tidal Variability in the Photobiology of the Seagrass Halophila johnsonii in a Riverine Versus Marine Habitat. Estuaries and Coasts 36(2):430-443.

Kahn, A. E., and M. J. Durako. 2008. Photophysiological responses of *Halophila johnsonii* to experimental hyposaline and hyper-CDOM conditions. Journal of Experimental Marine Biology and Ecology 367(2):230-235.

Kahn, A. E., and M. J. Durako. 2009. Photosynthetic tolerances to desiccation of the co-occurring seagrasses Halophila johnsonii and Halophila decipiens. Aquatic Botany 90(2):195-198.

Kahn, J., and M. Mohead. 2010. A Protocol for Use of Shortnose, Atlantic, Gulf, and Green Sturgeons.

Kahn, J. E., and coauthors. 2014. Atlantic sturgeon annual spawning run estimate in the Pamunkey River, Virginia. Transactions of the American Fisheries Society 143(6):1508-1514.

Kahnle, A. W., K. A. Hattala, and K. A. McKown. 2007. Status of Atlantic sturgeon of the Hudson River Estuary, New York, USA. American Fisheries Society Symposium 56:347-363.

Kahnle, A. W., and coauthors. 1998. Stock Status of Atlantic sturgeon of Atlantic Coast Estuaries. Atlantic States Marine Fisheries Comission.

Kashiwagi, T., T. Ito, and F. Sato. 2010. Occurences of reef manta ray, *Manta alfredi*, and giant manta ray, *M. birostris*, in Japan, examined by photographic records. Japanese Society for Elasmobranch Studies 46:20-27.

Kashiwagi, T., A. D. Marshall, M. B. Bennett, and J. R. Ovenden. 2011. Habitat segregation and mosaic sympatry of the two species of manta ray in the Indian and Pacific Oceans: *Manta alfredi* and *M. birostris*. Marine Biodiversity Records 4:1-8.

Keck, J., R. S. Houston, S. J. Purkis, and B. Riegl. 2005. Unexpectedly high cover of Acropora cervicornis on offshore reefs in Roatán (Honduras). Coral Reefs 24(3):509.

Keinath, J. A., and J. A. Musick. 1993. Movements and diving behavior of a leatherback turtle, *Dermochelys coriacea*. Copeia 1993(4):1010-1017.

Keller, J. M., P. D. McClellan-Green, J. R. Kucklick, D. E. Keil, and M. M. Peden-Adams. 2006. Effects of organochlorine contaminants on loggerhead sea turtle immunity: Comparison of a correlative field study and *in vitro* exposure experiments. Environmental Health Perspectives 114(1):70-76.

Keller, J. M., M. A. Stamper, J. R. Kucklick, and P. McClellan-Green. 2005. Assessment of immune function and presence of contaminants in the loggerhead sea turtle (*Caretta caretta*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-528, Miami, FL.

Kemp, D. W., and coauthors. 2011. Catastrophic mortality on inshore coral reefs of the Florida Keys due to severe low-temperature stress. Global Change Biology 17(11):3468-3477.

Kenworthy, W. J. 1993. The distribution, abundance, and ecology of *Halophila johnsonii* Eiseman in the lower Indian River, Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

Kenworthy, W. J. 1997a. An updated biological status review and summary of the proceedings of a workshop to review the biological status of the seagrass *Halophila johnsonii* Eiseman. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Beaufort, North Carolina.

Kenworthy, W. J. 1997b. An updated biological status review and summary of the proceedings of a workshop to review the biological status of the seagrass, *Halophila johnsonii* Eiseman. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Beaufort, NC.

Kenworthy, W. J. 2000. The role of sexual reproduction in maintaining populations of *Halophila decipiens*: Implications for the biodiversity and conservation of tropical seagrass ecosystems. Pacific Conservation Biology 5(4):260-268.

Kenworthy, W. J., and M. S. Fonseca. 1996. Light requirements of seagrasses *Halodule wrightii* and *Syringodium filiforme* derived from the relationship between diffuse light attenuation and maximum depth distribution. Estuaries and Coasts 19(3):740-750.

Kenworthy, W. J., M. S. Fonseca, P. E. Whitfield, and K. K. Hammerstrom. 2002. Analysis of seagrass recovery in experimental excavations and propeller-scar disturbances in the Florida Keys National Marine Sanctuary. Journal of Coastal Research (37):75-85.

Kenworthy, W. J., and D. E. Haunert (editors). 1991. The light requirements of seagrasses: Results and recommendations of a workshop. Proceedings of a workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-287.

Kenworthy, W. J., S. Wyllie-Echeverria, R. G. Coles, G. Pergent, and C. Pergent-Martini. 2006. Seagrass conservation biology: An interdisciplinary science for protection of the seagrass biome. Pages 595-623 *in* A. W. D. Larkum, R. J. Orth, and C. M. Duarte, editors. Seagrasses: Biology, Ecology and Conservation. Springer Netherlands.

Kieffer, M. C., and B. Kynard. 1996. Spawning of the shortnosesturgeon in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society 125(2):179-186.

King, T. L., B. A. Lubinski, and A. P. Spidle. 2001. Microsatellite DNA variation in Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) and cross-species amplification in the Acipenseridae. Conservation Genetics 2(2):103-119.

Kintisch, E. 2006. As the seas warm. Science 313(5788):776-779.

Kirsch, K. D., and coauthors. 2005. The Mini-312 Program - An expedited damage assessment and restoration process for seagrasses in the Florida Keys National Marine Sanctuary. Journal of Coastal Research (40):109-119.

Knowlton, N., J. L. Maté, H. M. Guzmán, R. Rowan Jr., and J. M. Jara. 1997. Direct evidence for reproductive isolation among the three species of the *Montastraea annularis* complex in Central America (Panamá and Honduras). Marine Biology 127(4):705-711.

Kubis, S. A., M. Chaloupka, L. M. Ehrhart, and M. J. Bresette. 2009. Growth rates of juvenile green turtles *Chelonia mydas* from three ecologically distinct foraging habitats along the east central coast of Florida, USA. Marine Ecology Progress Series 389:257-269.

Kunzelman, J. 2007. Southern Range, permanent transect implementation, summer sampling 2006. Report prepared for the Johnson's Seagrass Recovery Team. Florida Fish and Wildlife Conservation Commission, Saint Petersburg, FL.

Kynard, B. 1997. Life history, latitudinal patterns, and status of the shortnose sturgeon, *Acipenser brevirostrum*. Environmental Biology of Fishes 48(1-4):319-334.

Kynard, B., and M. Horgan. 2002. Ontogenetic behavior and migration of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, and shortnose sturgeon, *A. brevirostrum*, with notes on social behavior. Environmental Biology of Fishes 63(2):137-150.

Kynard, B., M. Kieffer, M. Burlingame, and M. Horgan. 1999. Studies on shortnose sturgeon: Final report. Northeast Utilities Service Company, Berlin, CT and the City of Holyoke, MA.

Ladich, F., and R. R. Fay. 2013. Auditory evoked potential audiometry in fish. Reviews in Fish Biology and Fisheries 23(3):317-364.

Ladich, F., and A. N. Popper. 2004. Parallel evolution in fish hearing organs. Pages 95-127 *in* G. A. Manley, R. R. Fay, and A. N. Popper, editors. Evolution of the Vertebrate Auditory System. Springer Handbook of Auditory Research, volume 22. Springer, New York, NY.

Lagueux, C. J. 2001. Status and distribution of the green turtle, *Chelonia mydas*, in the wider Caribbean region. Pages 32-35 *in* K. L. Eckert, and F. A. Abreu-Grobois, editors. Proceedings of the Regional Meeting: "Marine Turtle Conservation in the Wider Caribbean Region: A Dialogue for Effective Regional Management". WIDECAST, IUCN-MTSG, WWF, UNEP-CEP, St. Croix, U.S. Virgin Islands.

Lahanas, P. N., and coauthors. 1998. Genetic composition of a green turtle (*Chelonia mydas*) feeding ground population: Evidence for multiple origins. Marine Biology 130(3):345-352.

Laney, R. W., and coauthors. 2007. Distribution, habitat use, and size of Atlantic sturgeon captured during cooperative winter tagging cruises, 1988-2006. Amercian Fisheries Society Symposium 56:167-182.

Lapointe, B. E., P. J. Barile, and W. R. Matzie. 2004. Anthropogenic nutrient enrichment of seagrass and coral reef communities in the Lower Florida Keys: Discrimination of local versus regional nitrogen sources. Journal of Experimental Marine Biology and Ecology 308(1):23-58.

Laurent, L., and coauthors. 1998. Molecular resolution of marine turtle stock composition in fishery by-catch: A case study in the Mediterranean. Molecular Ecology 7:1529-1542.

Law Engineering and Environmental Services, I. 1998. Agitation Sled Dredging Water Quality Assessment Berth 1 and 2, Port of Wilmington.

Law, K. L., and coauthors. 2010. Plastic accumulation in the North Atlantic subtropical gyre. Science 329(5996):1185-1188.

Law, R. J., and coauthors. 1991. Concentrations of trace metals in the livers of marine mammals (seals, porpoises and dolphins) from waters around the British Isles. Marine Pollution Bulletin 22(4):183-191.

Lawson, J. 1709. A New Voyage to Carolina; Containing the Exact Description and Natural History of That Country: Together with the Present State Thereof. And a Journal of a Thousand Miles, Travel'd Thro' Several Nations of Indians. Giving a Particular Account of Their Customs, Manners, &c. Retrieved from the Library of Congress, Control Number 02018519, https://www.loc.gov/item/02018519/, London: s.n.

Lawson, J. M., and coauthors. 2017. Sympathy for the devil: a conservation strategy for devil and manta rays. PeerJ 5:e3027.

Lawson, J. M., and coauthors. 2016. Sympathy for the devil: A conservation strategy for devil and manta rays. PeerJ 5:e3027.

Learmonth, J. A., and coauthors. 2006. Potential effects of climate change on marine mammals. Oceanography and Marine Biology: An Annual Review 44:431-464.

Leland III, J. G. 1968. A survey of the sturgeon fishery of South Carolina. Bears Bluff Laboratories, Wadmalaw Island, SC.

Lenhardt, M. L. 2002. Sea turtle auditory behavior. The Journal of the Acoustical Society of America 112(5):2314-2314.

Levitan, D. R., N. D. Fogarty, J. F. Jara, K. E. Lotterhos, and N. Knowlton. 2011. Genetic, spatial, and temporal components of precise spawning synchrony in reef building corals of the *Montastraea annularis* species complex. Evolution 65(5):1254-1270.

Lezama, C. 2009. Impacto de la pesquería artesanal sobre la tortuga verde (*Chelonia mydas*) en las costas del Río de la Plata exterior. Universidad de la República, Montevideo, Uruguay.

Lidz, B. H., and D. G. Zawada. 2013. Possible return of *Acropora cervicornis* at Pulaski Shoal, Dry Tortugas National Park, Florida. Journal of Coastal Research 29(2):256-271.

Lighty, R. G., I. G. Macintyre, and R. Stuckenrath Jr. 1978. Submerged early Holocene barrier reef south-east Florida shelf. Nature 276:59-60.

Lighty, R. G., I. G. Macintyre, and R. Stuckenrath Jr. 1982. *Acropora palmata* reef framework: A reliable indicator of sea level in the western atlantic for the past 10,000 years. Coral Reefs 1(2):125-130.

Lima, E. H. S. M., M. T. D. Melo, and P. C. R. Barata. 2010. Incidental capture of sea turtles by the lobster fishery off the Ceará Coast, Brazil. Marine Turtle Newsletter 128:16-19.

Lindahl, U. 2003. Coral reef rehabilitation through transplantation of staghorn corals: Effects of artificial stabilization and mechanical damages. Coral Reefs 22(3):217-223.

Lirman, D. 2000. Fragmentation in the branching coral *Acropora palmata* (Lamarck): Growth, survivorship, and reproduction of colonies and fragments. Journal of Experimental Marine Biology and Ecology 251(1):41-57.

Lirman, D., and coauthors. 2010. A window to the past: documenting the status of one of the last remaining 'megapopulations' of the threatened staghorn coral Acropora cervicornis in the Dominican Republic. Aquatic Conservation: Marine and Freshwater Ecosystems 20(7):773-781.

Lirman, D., and coauthors. 2016. Summary report for Biscayne Bay. Florida Fish and Wildlife Research Institute, Technical Report TR-17, Version 2, St. Petersburg, FL.

Lirman, D., and coauthors. 2011. Severe 2010 cold-water event caused unprecedented mortality to corals of the Florida Reef Tract and reversed previous survivorship patterns. PLOS ONE 6(8):e23047.

Lloyd, B. D. 2003. Potential effects of mussel farming on New Zealand's marine mammals and seabirds: A discussion paper. Department of Conservation, Wellington, New Zealand.

Longwell, A. C., S. Chang, A. Hebert, J. B. Hughes, and D. Perry. 1992. Pollution and developmental abnormalities of Atlantic fishes. Environmental Biology of Fishes 35(1):1-21.

López-Barrera, E. A., G. O. Longo, and E. L. A. Monteiro-Filho. 2012. Incidental capture of green turtle (*Chelonia mydas*) in gillnets of small-scale fisheries in the Paranaguá Bay, Southern Brazil. Ocean and Coastal Management 60:11-18.

López-Mendilaharsu, M., and coauthors. 2006. Biología, ecología y etología de las tortugas marinas en la zona costera uruguaya. Bases para la conservación y el manejo de la costa uruguaya:11.

Lovell, J. M., M. M. Findlay, R. M. Moate, J. R. Nedwell, and M. A. Pegg. 2005. The inner ear morphology and hearing abilities of the Paddlefish (*Polyodon spathula*) and the Lake Sturgeon (*Acipenser fulvescens*). Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology 142(3):286-296.

Luksenburg, J. A., and E. C. M. Parsons. 2014. Attitudes towards marine mammal conservation issues before the introduction of whale-watching: A case study in Aruba (southern Caribbean). Aquatic Conservation: Marine and Freshwater Ecosystems 24(1):135-146.

Lundgren, I., and Z.-M. Hillis-Starr. 2008. Variation in *Acropora palmata* bleaching across benthic zones at Buck Island Reef National Monument (St. Croix, USVI) during the 2005 thermal stress event. Bulletin of Marine Science 83(3):441-451.

Lunz, K. S. 2013. Final Report Permit Number: FKNMS-2010-126-A3. Florida Fish and Wildlife Conservation Commission, St. Petersburg, FL.

Lutcavage, M. E., and P. L. Lutz. 1997. Diving physiology. Pages 277-295 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Lutcavage, M. E., P. Plotkin, B. Witherington, and P. L. Lutz. 1997. Human impacts of sea turtle survival. Pages 387-404 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Lutz, P. L., and A. Dunbar-Cooper. 1987. Variations in the blood chemistry of the loggerhead sea turtle, *Caretta caretta*. Fishery Bulletin 85(1):37-43.

Mac, M. J., and C. C. Edsall. 1991. Environmental contaminants and the reproductive success of lake trout in the Great Lakes: An epidemiological approach. Journal of Toxicology and Environmental Health 33(4):375-394.

Macintyre, I. G., and M. A. Toscano. 2007. The elkhorn coral *Acropora palmata* is coming back to the Belize Barrier Reef. Coral Reefs 26(4):757.

MacLeod, C. D., and coauthors. 2005. Climate change and the cetacean community of north-west Scotland. Biological Conservation 124(4):477-483.

Maguire, J.-J., M. Sissenwine, J. Csirke, and R. Grainger. 2006. The state of the world highly migratory, straddling and other high seas fish stocks, and associated species. Food and Agricultural Organization of the United Nations, FAO Fisheries Technical Paper No. 495, Rome, Italy.

Maharaj, A. M. 2004. A comparative study of the nesting ecology of the leatherback turtle *Dermochelys coriacea* in Florida and Trinidad. Master's Thesis. University of Central Florida, Orlando, FL.

Maine State Planning Office. 1993. Kennebec River Resource Management Plan: Balancing hydropower generation and other uses. Final Report to the Maine State Planning Office. Maine State Planning Office, Paper 78, Augusta, ME.

MantaMatcher. 2016. Manta Matcher - The Wildbook for Manta Rays.

Marcovaldi, N., B. B. Gifforni, H. Becker, F. N. Fiedler, and G. Sales. 2009. Sea turtle interactions in coastal net fisheries in Brazil. Western Pacific Regional Fishery Management Council, IUCN, Southeast Asian Fisheries Development Center, Indian Ocean - South-East Asian Marine Turtle MoU, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Honolulu, HI.

Marcy Jr., B. C., D. E. Fletcher, F. D. Martin, M. H. Paller, and M. J. M. Reichert. 2005. Fishes of the middle Savannah River Basin: With emphasis on the Savannah River Site. University of Georgia Press.

Marine Mammal Commission. 2007. Marine mammals and noise: A sound approach to research and management. A report to Congress from the Marine Mammal Commission. Marine Mammal Commission.

Márquez M., R. 1990. Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date. Food and Agricultural Organization of the United Nations, FAO Fisheries Synopsis No. 125, Rome, Italy.

Márquez M., R. 1994. Synopsis of biological data on the Kemp's ridley sea turtle, *Lepidochelys kempii* (Garman, 1880). Instituto Nacional de la Pesca, U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region and U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-343, Miami, FL.

Marshall, A., and coauthors. 2011. *Manta birostris*. The IUCN Red List of Threatened Species.

Marshall, A. D., L. J. V. Compagno, and M. B. Bennett. 2009. Redescription of the genus Manta with resurrection of *Manta alfredi* (Krefft, 1868) (Chondrichthyes; Myliobatoidei; Mobulidae). Zootaxa 2301:1-28.

Martin, B., J. MacDonnell, N. E. Chorney, and D. G. Zeddies. 2012. Sound Source 20 Verification of Fugro Geotechnical Sources: Final Report: Boomer, Sub-Bottom 21 Profiler, Multibeam Sonar, and the R/V Taku. JASCO Document 00413, Version 1.0 DRAFT. Technical report by JASCO Applied Sciences for Fugro GeoServices Inc. 31 pp.

Martínez, L. S., and coauthors. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology 6(1):70-78.

Matkin, C., and E. Saulitis. 1997. Killer whale *Orcinus orca*. Restoration Notebook, *Exxon Valdez* Oil Spill Trustee Council.

Matos, R. 1986. Sea turtle hatchery project with specific reference to the leatherback turtle (*Dermochelys coriacea*), Humacao, Puerto Rico 1986. Puerto Rico Department of Natural Resources, de Tierra, Puerto Rico.

Matta, M. B., C. Cairncross, and R. M. Kocan. 1997. Effect of a polychlorinated biphenyl metabolite on early life stage survival of two species of trout. Bulletin of Environmental Contamination and Toxicology 59(1):146-151.

Mayor, P. A., C. S. Rogers, and Z.-M. Hillis-Starr. 2006. Distribution and abundance of elkhorn coral, *Acropora palmata*, and prevalence of white-band disease at Buck Island Reef National Monument, St. Croix, US Virgin Islands. Coral Reefs 25(2):239-242.

McCauley, R. D., and coauthors. 2000. Marine seismic surveys: Analysis and propagation of air-gun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid Curtin University of Technology, Western Australia.

McCord, J. W., M. R. Collins, W. C. Post, and T. I. J. Smith. 2007. Attempts to develop an index of abundance for age-1 Atlantic sturgeon in South Carolina, USA. American Fisheries Society Symposium 56:397-403.

McDonald, D. L., and P. H. Dutton. 1996. Use of PIT tags and photoidentification to revise remigration estimates of leatherback turtles (*Dermochelys coriacea*) nesting in St. Croix, U.S. Virgin Islands, 1979-1995. Chelonian Conservation and Biology 2(2):148-152.

McDonald, M. 1887. Subsection 2. The rivers and sounds of North Carolina. Pages 625-637 *in* The Fisheries and Fishery Industries of the United States Vol.1, Section V. History and Methods of the Fisheries, Part XII. The River Fisheries of the Atlantic States. U.S. Commission of Fish and Fisheries, Government Printing Office, Washington, D.C.

McKenna, M. F., D. Ross, S. M. Wiggins, and J. A. Hildebrand. 2012. Underwater radiated noise from modern commercial ships. Journal of the Acoustical Society of America 131(1):92-103.

McKenna, M. F., S. M. Wiggins, and J. A. Hildebrand. 2013. Relationship between container ship underwater noise levels and ship design, operational and oceanographic conditions. Scientific Reports 3.

McKeown, B. A. 1984. Fish Migration. Timber Press, Portland, Oregon.

McMahon, C. R., and G. C. Hays. 2006. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology 12(7):1330-1338.

McMichael, E., R. R. Carthy, and J. A. Seminoff. 2003. Evidence of homing behavior in juvenile green turtles in the northeastern Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

McMillan, C., and S. C. Williams. 1980. Systematic implications of isozymes in Halophila section Halophila. Aquatic Botany 9:21-31.

Medeiros, A. M., O. J. Luiz, and C. Domit. 2015. Occurrence and use of an estuarine habitat by giant manta ray *Manta birostris*. Journal of Fish Biology 86(6):1830-1838.

Mège, P., N. V. Schizas, J. Garcia Reyes, and T. Hrbek. 2015. Genetic seascape of the threatened Caribbean elkhorn coral, *Acropora palmata*, on the Puerto Rico Shelf. Marine Ecology 36(2):195-209.

Meyer, M., and A. N. Popper. 2002. Hearing in primitive fish: Brainstem responses to pure tone stimuli in the lake sturgeon, Acipenser fulvescens. Association for Research in Otolaryngology 25:11-12.

Meylan, A. B., B. A. Schroeder, and A. Mosier. 1994. Marine turtle nesting activity in the state of Florida, 1979-1992. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-351, Miami, FL.

Meylan, A. B., B. A. Schroeder, and A. Mosier. 1995. Sea turtle nesting activity in the state of Florida, 1979-1992. State of Florida, Department. of Environmental Protection, Florida Marine Research Institute, 0095-0157, Saint Petersburg, FL.

Meylan, A. B., B. E. Witherington, B. Brost, R. Rivera, and P. S. Kubilis. 2006. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of *Caretta*, *Chelonia*, and *Dermochelys*, Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation edition. International Sea Turtle Society, Athens, Greece.

Meylan, P. A., A. B. Meylan, and J. A. Gray. 2011. The ecology and migrations of sea turtles 8. Tests of the developmental habitat hypothesis. Bulletin of the American Museum of Natural History 357:1-70.

Milessi, A. C., and M. C. Oddone. 2003. Primer registro de *Manta birostris* (Donndorff 1798) (Batoidea: Mobulidae) en el Rio de La Plata, Uruguay. Gayana 67(1):126-129.

Miller, J. D. 1997. Reproduction in sea turtles. Pages 51-81 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Miller, M. H., and C. Klimovich. 2017. Endangered Species Act status review report: Giant manta ray (*Manta birostris*) and reef manta ray (*Manta alfredi*). U.S. Department of Commerce, National Oceanic and Atmoshperic Administration, National Marine Fisheries Servcie, Office of Protected Resources, Silver Spring, MD.

Miller, M. W., I. B. Baums, and D. E. Williams. 2007. Visual discernment of sexual recruits is not feasible for *Acropora palmata*. Marine Ecology Progress Series 335:227-231.

Miller, M. W., K. Kerr, and D. E. Williams. 2016. Reef-scale trends in Florida *Acropora* spp. abundance and the effects of population enhancement. PeerJ 4:e2523.

Miller, S. L., M. Chiappone, L. M. Rutten, and D. W. Swanson. 2008. Population status of *Acropora* corals in the Florida Keys, Ft. Lauderdale, FL.

Miller, T., and G. Shepherd. 2011. Summary of discard estimates for Atlantic sturgeon. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Science Center, Greater Atlantic Regional Fishiers Office, Northeast Fisheries Science Center, Population Dynamics Branch.

Milton, S. L., and P. L. Lutz. 2003. Physiological and genetic responses to environmental stress. Pages 163-197 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, Vol. II. CRC Press, Boca Raton, FL.

Miranda, L. E., and K. J. Kilgore. 2013. Entrainment of shovelnose sturgeon by towboat navigation in the Upper Mississippi River. Journal of Applied Ichthyology 29(2):316–322.

Mo, C. L. 1988. Effect of bacterial and fungal infection on hatching success of olive ridley sea turtle eggs. Report to the U.S. World Wildlife Fund.

Moncada-Gavilán, F. B. 2001. Status and distribution of the loggerhead turtle, *Caretta caretta*, in the wider Caribbean region. Pages 36-40 *in* K. L. Eckert, and F. A. Abreu-Grobois, editors. Proceedings of the Regional Meeting: "Marine Turtle Conservation in the Wider Caribbean Region - A Dialogue for Effective Regional Management". WIDECAST, IUCN-MTSG, WWF, UNEP-CEP, St. Croix, U.S. Virgin Islands.

Moncada, F. B., and coauthors. 2010. Movement patterns of loggerhead turtles *Caretta caretta* in Cuban waters inferred from flipper tag recaptures. Endangered Species Research 11(1):61-68.

Moncada, F. B., and coauthors. 2006. Movement patterns of green turtles (*Chelonia mydas*) in cuba and adjacent caribbean waters inferred from flipper tag recaptures. Journal of Herpetology 40(1):22-34.

Moncheva, S. P., and L. T. Kamburska. 2002. Plankton stowaways in the Black Sea - Impacts on biodiversity and ecosystem health. Commission Internationale pour l'Exploration Scientifique de la mer Méditerranée (CIESM) Workshop Monographs 20, Istanbul, Turkey.

Mongillo, T. M., and coauthors. 2012. Predicted polybrominated diphenyl ether (PBDE) and polychlorinated biphenyl (PCB) accumulation in southern resident killer whales. Marine Ecology Progress Series 453:263-277.

Monzón-Argüello, C., and coauthors. 2010. Evidence from genetic and Lagrangian drifter data for transatlantic transport of small juvenile green turtles. Journal of Biogeography 37(9):1752-1766.

Moore, A., and C. P. Waring. 2001. The effects of a synthetic pyrethroid pesticide on some aspects of reproduction in Atlantic salmon (*Salmo salar* L.). Aquatic Toxicology 52(1):1-12.

Moore, A. B. M. 2012. Records of poorly known batoid fishes from the north-western Indian Ocean (Chondrichthyes: Rhynchobatidae, Rhinobatidae, Dasyatidae, Mobulidae). African Journal of Marine Science 34(2):297-301.

Morris, L. J., L. Hall, R. Chamberlain, and C. Jacoby. 2018. Summary report for the Northern Indian River Lagoon. Florida Fish and Wildlife Research Institute, Technical Report 17, St. Petersburg, FL.

Moser, M. L., J. B. Bichy, and S. B. Roberts. 1998. Sturgeon distibution in North Carolina. University of North Carolina Wilmington, Center for Marine Science Research, Wilmington, NC.

Moser, M. L., and S. W. Ross. 1995. Habitat use and movements of shortnose and Atlantic sturgeons in the lower Cape Fear River, North Carolina. Transactions of the American Fisheries Society 124(2):225.

Mourier, J. 2012. Manta rays in the Marquesas Islands: First records of *Manta birostris* in French Polynesia and most easterly location of *Manta alfredi* in the Pacific Ocean, with notes on their distribution. Journal of Fish Biology 81(6):2053-2058.

Mrosovsky, N., G. D. Ryan, and M. C. James. 2009. Leatherback turtles: The menace of plastic. Marine Pollution Bulletin 58(2):287-289.

Muller, E. M., C. S. Rogers, A. S. Spitzack, and R. van Woesik. 2008. Bleaching increases likelihood of disease on *Acropora palmata* (Lamarck) in Hawksnest Bay, St. John, US Virgin Islands. Coral Reefs 27(1):191-195.

Muller, E. M., C. S. Rogers, and R. van Woesik. 2014. Early signs of recovery of *Acropora palmata* in St. John, US Virgin Islands. Marine Biology 161(2):359-365.

Mumby, P. J., and A. R. Harborne. 2010. Marine reserves enhance the recovery of corals on Caribbean reefs. PLOS ONE 5(1):e8657.

Murawski, S. A., and A. L. Pacheco. 1977. Biological and fisheries data on Atlantic sturgeon, *Acipenser oxyrhynchus* (Mitchill). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Center, Sandy Hook Laboratory, Technical Series Report No. 10, Highlands, NJ.

Murdoch, P. S., J. S. Baron, and T. L. Miller. 2000. Potential effects of climate change of surface water quality in North America. JAWRA Journal of the American Water Resources Association 36(2):347-366.

Murphy, T. M., and S. R. Hopkins. 1984. Aerial and ground surveys of marine turtle nesting beaches in the southeast region, U.S.: Final report to the National Marine Fisheries Service. LaMER, NMFS Contract Number NA83-GA-C-00021, Green Pond, SC.

Musick (editor), J. A. 1999. Life in the Slow Lane: Ecology and conservation of long-lived marine animals, volume 23. American Fisheries Society, Bethesda, MD.

Musick, J. A., and C. J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pages 137-163 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, FL.

Myrberg Jr., A. A. 2001. The acoustical biology of elasmobranchs. Environmental Biology of Fishes 60(1):31-45.

Naro-Maciel, E., J. H. Becker, E. H. S. M. Lima, M. Â. Marcovaldi, and R. DeSalle. 2007. Testing dispersal hypotheses in foraging green sea turtles (*Chelonia mydas*) of Brazil. Journal of Heredity 98(1):29-39.

Naro-Maciel, E., and coauthors. 2012. The interplay of homing and dispersal in green turtles: A focus on the southwestern Atlantic. Journal of Heredity 103(6):792-805.

NAST. 2000. Climate change impacts on the United States: the potential consequences of climate variability and change. US Global Change Research Program, Washington D.C. National Assessment Synthesis Team.

National Assessment Synthesis Team. 2000. Climate change impacts on the United States: The potential consequences of climate variability and change. U.S. Global Change Research Program, National Assessment Synthesis Team (NAST), Washington DC.

National Drought Mitigation Center. 2018. United States Drought Monitor. National Drought Mitigation Center at the University of Nebraska-Lincoln, U.S. Department of Agriculture, and National Oceanic and Atmospheric Administration, Lincoln, NE.

National Research Council. 1990a. Decline of the Sea Turtles: Causes and Prevention. National Academy Press, Washington, D.C.

National Research Council. 1990b. Sea turtle mortality associated with human activities. Pages 74-117 *in* National Research Council Committee on Sea Turtle Conservation, editor. Decline of the Sea Turtles: Causes and Prevention. National Academy Press, Washington, D.C.

National Research Council. 2003. Ocean Noise and Marine Mammals. The National Academies Press, Washington, D.C.

National Research Council. 2005. Marine Mammal Populations and Ocean Noise: Determining when noise causes biologically significant effects. The National Academes Press, Washington, D.C.

National Research Council. 2008. Tackling marine debris in the 21st century. The National Academies Press, Washington, D.C.

NCSPA. 2017. The Biological Assessment for Agitation and Water Injection Dredging Port of Morehead City, North Carolina State Ports Authority, prepared by Dial Cordy and Associates Inc.

Nelson, D. S., and coauthors. 2016. Predicting dredging-associated effects to coral reefs in Apra Harbor, Guam - Part 2: Potential coral effects. Journal of Environmental Management 168:111-122.

Newell, R. C., L. J. Seiderer, and D. R. Hitchcock. 1998. The impact of dredging works in coastal waters: A review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. Oceanography and Marine Biology: An Annual Review 36:127-178.

Niklitschek, E. J., and D. H. Secor. 2005. Modeling spatial and temporal variation of suitable nursery habitats for Atlantic sturgeon in the Chesapeake Bay. Estuarine, Coastal and Shelf Science 64(1):135-148.

Niklitschek, E. J., and D. H. Secor. 2009a. Dissolved oxygen, temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: I. Laboratory results. Journal of Experimental Marine Biology and Ecology.

Niklitschek, E. J., and D. H. Secor. 2009b. Dissolved oxygen, temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: I. Laboratory results. Journal of Experimental Marine Biology and Ecology 381(Supplement 1):S150-S160.

Niklitschek, E. J., and D. H. Secor. 2009c. Dissolved oxygen, temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: II. Model development and testing. Journal of Experimental Marine Biology and Ecology 381(Supplement 1):S161-S172.

Niklitschek, E. J., and D. H. Secor. 2010. Experimental and field evidence of behavioural habitat selection by juvenile Atlantic *Acipenser oxyrinchus oxyrinchus* and shortnose *Acipenser brevirostrum* sturgeons. Journal of Fish Biology 77(6):1293-1308.

NIOSH. 1998. Criteria for a recommended standard: Occupational noise exposure; Revised criteria 1998. U.S. Department of Health and Human Services, Public Health Service, Centers for Disease Control and Prevention National Institute for Occupational Safety and Health, DHHS (NIOSH) Publication No. 98–126, Cincinnati, OH.

NMFS. 1991. Dredging of channels of the southeastern U.S. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, 910049, Saint Petersburg, FL.

NMFS. 1995. Biologincal Opinion on hopper dredging of channels and beach nourishment activities in the southeastern United States from North Carolina through Florida East Coast. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, 950055, Saint Petersburg, FL.

NMFS. 1997. Biological Opinion on continued hopper dredging of channels and borrow areas in the southeastern United States. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 1998. Final recovery plan for the shortnose sturgeon (*Acipenser brevirostrum*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS. 1999. The use of the sidecast dredges FRY, MERRITT and SCHWEIZER, and the split-hull hopper dredge CURRITUCK. U.S. Department of Commerce, National Oceanic and

Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 2000. Status review of smalltooth sawfish, *Pristis pectinata*. NOAA Fisheries, Southeast Regional Office, St. Petersburg, FL.

NMFS. 2001a. Biological Opinion: SAJ-2005-972 Florida Inland Navigation District Regional General Permit SAJ-93 to maintenance dredge AIWW, ICW, and Okeechobee Waterway along the east coast of Florida, SER-2000-1199, St Petersburg, FL.

NMFS. 2001b. Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the western North Atlantic. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-455, Miami, FL.

NMFS. 2002a. Biological Opinion on shrimp trawling in the southeastern United States, under the sea turtle conservation regulations and as managed by the fishery management plans for shrimp in the South Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 2002b. Final recovery plan for Johnson's seagrass (*Halophila johnsonii* Eiseman). U.S. Deparment of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS. 2007a. Endangered Species Act 5-year review: Johnson's seagrass (*Halophila johnsonii* Eiseman). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS. 2007b. Gulf of Mexico Regional Biological Opinion on Hopper Dredging (Revision 2 to GOM RBO). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Region, Protected Resources Division, SER-2006-2953, Saint Petersburg, FL.

NMFS. 2008. Sea turtle research techniques manual. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-579, Miami, FL.

NMFS. 2009a. An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, PRD-08/09-14, Miami, FL.

NMFS. 2009b. Biological Opinion for the Dade County Beach Erosion Control Project, Contract "E". U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2009-00879, Saint Petersburg, FL.

NMFS. 2009c. Smalltooth sawfish recovery plan (*Pristis pectinata*). U.S. Departmet of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS. 2010a. A Biological Assessment of Shortnose Sturgeon (*Acipenser brevirostrum*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Woods Hole, Massachusetts.

NMFS. 2010b. Biological assessment of Shortnose sturgeon (*Acipenser brevirostrum*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Woods Hole, MA.

NMFS. 2010c. Biological Opinion: Savannah Harbor Federal Navigation Project - dredging: channel widening and deepening for Post-Panamax vessels, St Petersburg, FL.

NMFS. 2010d. Smalltooth sawfish 5-year review: summary and evaluation. NOAA Fisheries, Southeast Regional Office, St. Petersburg, FL.

NMFS. 2010e. Smalltooth sawfish (*Pristis pectinata* Latham) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, St. Petersburg, FL.

NMFS. 2011a. Biological Opinion on the Miami Harbor Navigation Project in Miami-Dade County, Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2011-29, Saint Petersburg, FL.

NMFS. 2011b. Informal Concurrence Letter for Hollywood and Hallendale beach renourishment. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, SER-2011-04723 and SER-2011-00494, Saint Petersburg, FL.

NMFS. 2011c. Preliminary summer 2010 regional abundance estimate of loggerhead turtles (*Caretta caretta*) in northwestern Atlantic Ocean continental shelf waters. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Reference Document 11-03, Woods Hole, MA.

NMFS. 2013. Reinitiation - Batched Biological Opinion for seven New England FMPs. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Protected Resources Division, NER-2012-1956, Gloucester, MA.

NMFS. 2014. Informal Concurrence Letter for batch of 2 beach nourishment consultations in Palm Beach County, Florida. U.S. Department of Commerce, National Oceanic and

Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 2015a. Biological Opinion on U.S. Navy Northwest training and testing activities and associated NMFS regulations and letters of authorization. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, FPR-2015-9110, Silver Spring, MD.

NMFS. 2015b. Programmatic Biological Opinion on live rock and marine bivalve aquaculture in the state of Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2014-13378, Saint Petersburg, FL.

NMFS. 2015c. Recovery outline: Pillar coral, rough cactus coral, lobed star coral, mountainous star coral, boulder star coral. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Saint Petersburg, FL.

NMFS. 2016a. Biological Opinion on the continued authorization of NMFS' integrated fisheries independent monitoring activities in the Southeast. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, SER-2009-7541, Saint Petersburg, FL.

NMFS. 2016b. Examination of sedimentation impacts to coral reef along the Port of Miami Entrance Channel, December 2015. U.S. Department of Commerce, National Oceanic and Atmoshperic Administration, National Marine Fisheries Service, Southeast Regional Office, Final Report, April 2016, Saint Petersburg, FL.

NMFS. 2017a. Biological Opinion on CENAP-OP-R- 2016-0181-39 DRP Gibbstown Shipping Terminal and Logistic Center. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, NER-2017-14371, Gloucester, MA.

NMFS. 2017b. Biological Opinion on the authorization of minor in-water activities throughout the geographic area of jurisdiction of the U.S. Army Corps of Engineers Jacksonville District, including Florida and the U.S. Caribbean (JAXBO). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division, SER-2015-17616, Saint Petersburg, FL.

NMFS. 2017c. Biological Opinion on U.S. Navy Gulf of Alaska training activities and the National Marine Fisheries Service's proposal to issue regulations and letters of authorization for incidental take of marine mammals pursuant to the Marine Mammal Protection Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, FPR-2015-9118, Silver Springs, MD.

NMFS. 2017d. Biological Opinion on United States Navy's surveillance towed array sensor system low frequency active sonar routine training, testing, and military operations from August 2017 through August 2022. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, FPR-2016-9181, Silver Spring, MD.

NMFS. 2017e. Process for determining post-interaction mortality determinations of sea turtles bycaught in trawl, net, and pot/trap fisheries. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, NMFSPI 02-110-21, Silver Spring, MD.

NMFS. 2017f. Recovery Outline - Atlantic Sturgeon - Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic Distinct Population Segments.

NMFS. 2018a. 2018 Revision to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0): Underwater thresholds for onset of permanent and temporary threshold shifts. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, NOAA Technical Memorandum NMFS-OPR-59, Silver Spring, MD.

NMFS. 2018b. Biological Opinion on the Deepening & Maintenance of the Deleware River Federal Navigation Channel , NER-2018-15005.

NMFS. 2018c. Biological Opinion: James River Federal Navigation Project (NER-2018-15090, GARFO-2018-00105).

NMFS. 2018d. Reinitiation - ACOE NAO Sandbridge Beach Nourishment, VA Hurricane Protection Project, Thimble Shoals, Atlantic Ocean Channel, York Spit, York River Entrance, Cape Henry. U.S. Department of Commerce, National Oceanic and Atmoshperic Administration, National Marine Fisheries Service, Northeast Region, NER-2012-1587, Gloucester, MA.

NMFS. 2018e. Smalltooth sawfish 5-year review: summary and evaluation. NOAA Fisheries, Southeast Regional Office
St. Petersburg, FL.

NMFS. 2019a. Giant manta ray recovery outline. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS. 2019b. Sand survey activities for BOEM's Marine Minerals Program: Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, NER-2018-15093, Gloucester, MA.

NMFS. 2020a. Biological Opinion for the Re-licensing of the South Carolina Public Service Authority Hydroelectric Project (FERC #199-205), SERO-2018-00325

NMFS. 2020b. Section 7 Effect Analysis: Turbidity in the Greater Atlantic Region. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Greater Atlantic Regional Fisheries Office, Gloucester, MA.

NMFS, USFWS, and SEMARNAT. 2011. Bi-national recovery plan for the Kemp’s ridley sea turtle (*Lepidochelys kempii*), Second revision. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 1991. Recovery plan for U.S. population of Atlantic green turtle (*Chelonia mydas*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Washington, D.C.

NMFS and USFWS. 1992. Recovery plan for the leatherback turtles *Dermochelys coriacea* in the U.S. Caribbean, Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Washington, D.C.

NMFS and USFWS. 1993. Recovery plan for the hawksbill turtle *Eretmochelys imbricata* in the U.S. Caribbean, Atlantic and Gulf of Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Saint Petersburg, FL.

NMFS and USFWS. 1995. Status reviews for sea turtles listed under the Endangered Species Act of 1973. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 1998. Recovery plan for U.S. Pacific populations of the leatherback turtle (*Dermochelys coriacea*). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Department of the Interior, U.S. Fish and Wildlife Service, Silver Spring, MD.

NMFS and USFWS. 2007a. Green sea turtle (*Chelonia mydas*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2007b. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Department of the Interior, U.S. Fish and Wildlife Service, Silver Spring, MD.

NMFS and USFWS. 2007c. Loggerhead sea turtle (*Caretta caretta*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Department of the Interior, U.S. Fish and Wildlife Service, Silver Spring, MD.

NMFS and USFWS. 2008. Recovery plan for the Northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*), Second revision. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS. 2013. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fiseries Service, Silver Spring, MD.

NOAA. 2012. Understanding Climate. http://www.climate.gov/#understandingClimate.

NOAA. 2018a. Climate at a Glance: National Time Series. National Centers for Environmental Information, published November 2019, retrieved on November 18, 2019 from https://www.ncdc.noaa.gov/cag/.

NOAA. 2018b. Status of Puerto Rico's coral reefs in the aftermath of Hurricanes Irma and Maria: Assessment report submitted by NOAA to the FEMA Natural and Cultural Resources Recovery Support Function. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Noriega, R., J. M. Werry, W. Sumpton, D. Mayer, and S. Y. Lee. 2011. Trends in annual CPUE and evidence of sex and size segregation of *Sphyrna lewini*: Management implications in coastal waters of northeastern Australia. Fisheries Research 110(3):472-477.

Northcote, T. G. 1978. Migratory strategies and production of freshwater fishes. Pages 326-359 *in* S. Gerking, editor. Ecology of freshwater fish production. John Wiley and Sons, New York, NY.

Northwest Atlantic Leatherback Working Group. 2018. Northwest Atlantic leatherback turtle (*Dermochelys coriacea*) status assessment. Conservation Science Partners and the Wider Caribbean Sea Turtle Conservation Network (WIDECAST), WIDECAST Technical Report No. 16, Godfrey, IL.

Notarbartolo di Sciara, G., and E. V. Hillyer. 1989. Mobulid rays off eastern Venezuela (Chondrichthyes, Mobulidae). Copeia (3):607-614.

Nowacek, D. P., L. H. Thorne, D. W. Johnston, and P. L. Tyack. 2007. Responses of cetaceans to anthropogenic noise. Mammal Review 37(2):81-115.

O'Hara, J., and J. R. Wilcox. 1990. Avoidance responses of loggerhead turtles, *Caretta caretta*, to low frequency sound. Copeia 1990(2):564-567.

O'Malley, M. P., K. Lee-Brooks, and H. B. Medd. 2013. The global economic impact of manta ray watching tourism. PLOS ONE 8(5):e65051.

Ogren, L. H. 1989. Distribution of juvenile and subadult Kemp's ridley sea turtles: Preliminary results from 1984-1987 surveys. Texas A&M University Sea Grant College Program, TAMU-SG-89-105, Galveston, TX.

Ogren, L. H., and C. McVea Jr. 1982. Apparent hibernation by sea turtles in North American waters. Smithsonian Institute Press, Washington, D.C.

Oliver, S., M. Braccini, S. J. Newman, and E. S. Harvey. 2015. Global patterns in the bycatch of sharks and rays. Marine Policy 54:86-97.

Orlando, B., E. Anderson, and L. A. Yarbro. 2016. Summary report for Lake Worth Lagoon. Florida Fish and Wildlife Research Institute, Technical Report TR-17, Version 2, St. Petersburg, FL.

Orlando, S. P., Jr. , and coauthors. 1994. Salinity Characteristics of South Atlantic Estuaries. NOAA, Office of Ocean Resources Conservation and Assessment, Silver Spring, MD.

Oros, J., O. M. Gonzalez-Diaz, and P. Monagas. 2009. High levels of polychlorinated biphenyls in tissues of Atlantic turtles stranded in the Canary Islands, Spain. Chemosphere 74(3):473-478.

Oviatt, C. A., C. D. Hunt, G. A. Vargo, and K. W. Kopchynski. 1981. Simulation of a storm event in marine microcosms. Journal of Marine Research 39(4):605-626.

Paladino, F. V., M. P. O'Connor, and J. R. Spotila. 1990. Metabolism of leatherback turtles, gigantothermy, and thermoregulation of dinosaurs. Nature 344:858-860.

Palmer, M. A., and coauthors. 2008. Climate change and the world's river basins: Anticipating management options. Frontiers in Ecology and the Environment 6(2):81-89.

Parks, S. E., D. R. Ketten, J. T. O'Malley, and J. Arruda. 2007. Anatomical predictions of hearing in the North Atlantic right whale. The Anatomical Record: Advances in Integrative Anatomy and Evolutionary Biology 290(6):734-744.

Parsley, M. J., N. D. Popoff, and J. G. Romine. 2011. Short-term response of subadult white sturgeon to hopper dredge disposal operations. North American Journal of Fisheries Management 31(1):1-11.

Patenaude, N. J., and coauthors. 2002. Aircraft sound and disturbance to bowhead and beluga whales during spring migration in the Alaskan Beaufort Sea. Marine Mammal Science 18(2):309-335.

Patrício, A. R., X. Velez-Zuazo, C. E. Diez, R. P. van Dam, and A. M. Sabat. 2011. Survival probability of immature green turtles in two foraging grounds at Culebra, Puerto Rico. Marine Ecology Progress Series 440:217-227.

Pershing, A. J., and coauthors. 2015. Slow adaptation in the face of rapid warming leads to collapse of the Gulf of Maine cod fishery. Science 350(6262):809-812.

Peterson, D., and coauthors. 2008. Annual run size and genetic characteristics of Atlantic sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 137:393-401.

Peterson, D. L., and M. S. Bednarski. 2013. Abundance and size structure of Shortnose Sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 142(5):1444-1452.

Peterson, D. L., and D. J. Farrae. 2011. Evidence of metapopulation dynamics in Shortnose Sturgeon in the southern part of their range. Transactions of the American Fisheries Society 140(6):1540-1546.

Philbrick, C. T., and D. H. Les. 1996. Evolution of aquatic angiosperm reproductive systems: What is the balance between sexual and asexual reporduction in aquatic angiosperms? BioScience 46(11):813-826.

Phillips, B. E., S. A. Cannizzo, M. H. Godfrey, B. A. Stacy, and C. A. Harms. 2015. Exertional myopathy in a juvenile green sea turtle (*Chelonia mydas*) entangled in a large mesh gillnet. Case Reports in Veterinary Medicine 2015:6.

Pike, D. A., R. L. Antworth, and J. C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the loggerhead sea turtle, *Caretta caretta*. Journal of Herpetology 40(1):91-94.

Plotkin, P. T. 2003. Adult migrations and habitat use. Pages 225-241 *in* P. L. Lutz, J. A. Musick, and J. Wyneken, editors. The Biology of Sea Turtles, Vol. II. CRC Press, Boca Raton, FL.

Poiner, I. R., and R. Kennedy. 1984. Complex patterns of change in the macrobenthos of a large sandbank following dredging. Marine Biology 78(3):335-352.

Pollock, F. J., and coauthors. 2014. Sediment and turbidity associated with offshore dredging increase coral disease prevalence on nearby reefs. PLOS ONE 9(7):e102498.

Polyakov, I. V., V. A. Alexeev, U. S. Bhatt, E. I. Polyakova, and X. Zhang. 2009. North Atlantic warming: Patterns of long-term trend and multidecadal variability. Climate Dynamics 34(2-3):439-457.

Popper, A. N., and coauthors. 2014. Sound Exposure Guidelines for Fishes and Sea Turtles: A technical report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI, ASA S3/SC1. 4 TR-2014. Springer, New York, NY.

Popper, A. N., and C. R. Schilt. 2008. Hearing and ccoustic behavior: Basic and applied considerations. Pages 17-48 *in* J. F. Webb, R. R. Fay, and A. N. Popper, editors. Fish Bioacoustics. Springer Handbook of Auditory Research, volume 32. Springer, New York NY.

Porter, J. W., and coauthors. 2001. Patterns of spread of coral disease in the Florida Keys. Hydrobiologia 460(1):1-24.

Porter, J. W., and coauthors. 2012. Catastrophic loss of *Acropora palmata* in the Florida Keys: Failure of the 'Sorcerer's Apprentice Effect' to aid recovery following the 2005 Atlantic hurricane season. James Cook University, Townsville, Queensland, Australia.

Posluszny, U., and P. B. Tomlinson. 1991. Shoot organization in the seagrass *Halophila* (Hydrocharitaceae). Canadian Journal of Botany 69(7):1600-1615.

Post, B., T. Darden, D. Peterson, M. Loeffler, and C. Collier. 2014a. Research and Management of Endangered and Threatened Species in the Southeast: Riverine Movements of Shortnose and Atlantic sturgeon. South Carolina Department of Natural Resources.

Post, G. W. 1987. Textbook of Fish Health, Revised and Expanded. TFH Publications, NJ.

Post, W. C., T. Darden, D. L. Peterson, M. Loeffler, and C. Collier. 2014b. Research and management of endangered and threatened species in the southeast: riverine movements of shortnose and Atlantic sturgeon. South Carolina Department of Natural Resources, Project NA10NMF4720036, Charleston, SC.

Post, W. C., S. C. Holbrook, and A. Watford. 2017a. Distribution and movement of shortnose sturgeon, and continued monitoring and maintenance of an existing acoustic receiver array. South Carolina Department of Natural Resources.

Post, W. C., S. C. Holbrook, and A. Watford. 2017b. Distribution and movement of shortnose sturgeon, and continued monitoring and maintenance of an existing acoustic receiver array.

Pottle, R., and M. J. Dadswell. 1979. Studies on larval and juvenile shortnose sturgeon (Acipenser brevirostrum). Report to the Northeast Utilities Service Company, Hartford, Connecticut.

Poulakis, G., P. Stevens, A. A. Timmers, T. R. Wiley, and C. Simpfendorfer. 2011a. Abiotic affinities and spatiotemporal distribution of the endangered smalltooth sawfish, *Pristis pectinata*, in a south-western Florida nursery. Marine and Freshwater Research 62:1165-1177.

Poulakis, G. R. 2012. Distribution, habitat use, and movements of juvenile smalltooth sawfish, *Pristis pectinata*, in the Charlotte Harbor estuarine system, Florida. Dissertation. Florida Institute of Technology, Melbourne, FL.

Poulakis, G. R., and J. C. Seitz. 2004. Recent occurrence of the smalltooth sawfish, *Pristis pectinata* (Elasmobranchiomorphi: Pristidae), in Florida Bay and the Florida Keys, with comments on sawfish ecology. Florida Scientist 67(1):27-35.

Poulakis, G. R., P. W. Stevens, A. A. Timmers, C. J. Stafford, and C. A. Simpfendorfer. 2013. Movements of juvenile endangered smalltooth sawfish, *Pristis pectinata*, in an estuarine river system: use of non-main-stem river habitats and lagged responses to freshwater inflow-related changes. Environmental Biology of Fishes 96(6):763-778.

Poulakis, G. R., P. W. Stevens, A. A. Timmers, T. R. Wiley, and C. A. Simpfendorfer. 2011b. Abiotic affinities and spatiotemporal distribution of the endangered smalltooth sawfish, *Pristis pectinata*, in a south-western Florida nursery. Marine and Freshwater Research 62(10):1165-1177.

Poulakis, G. R., and coauthors. 2017. Sympatric elasmobranchs and fecal samples provide insight into the trophic ecology of the smalltooth sawfish. Endangered Species Research 32:491-506.

Precht, W. F., and R. B. Aronson. 2004. Climate flickers and range shifts of reef corals. Frontiers in Ecology and the Environment 2(6):307-314.

Precht, W. F., B. E. Gintert, M. L. Robbart, R. Fura, and R. van Woesik. 2016. Unprecedented disease-related coral mortality in southeastern Florida. Scientific Reports 6:31374.

Price, C. S., and J. A. Morris Jr. 2013. Marine cage culture and the environment: Twenty-first century science informing a sustainable industry. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, NOAA Technical Memorandum NOS NCCOS 164.

Price, C. S., and coauthors. 2017. Protected species and marine aquaculture interactions. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, NOAA Technical Memorandum NOS NCCOS 211.

Pritchard, P. C. H. 1969. The survival status of ridley sea-turtles in America. Biological Conservation 2(1):13-17.

Pritchard, P. C. H., and P. Trebbau. 1984. The Turtles of Venezuela, Contributions to herpetology edition. Society for the Study of Amphibians and Reptiles.

Prohaska, B. K., and coauthors. 2018a. Physiological stress in the smalltooth sawfish: Effects of ontogeny, capture method, and habitat quality. Endangered Species Research 36.

Prohaska, B. K., and coauthors. 2018b. Physiological stress in the smalltooth sawfish: Effects of ontogeny, capture method, and habitat quality. Endangered Species Research 36:121-135.

Prosdocimi, L., V. González Carman, D. A. Albareda, and M. I. Remis. 2012. Genetic composition of green turtle feeding grounds in coastal waters of Argentina based on mitochondrial DNA. Journal of Experimental Marine Biology and Ecology 412:37-45.

Pyzik, L. a., J. Caddick, and P. Marx. 2004. Chesapeake Bay: Introduction to an ecosystem U.S. Environmental Protection Agency, EPA 903-R-04-003, CBP/TRS 232/00.

Quinn, N., and B. Kojis. 2008. The recent collapse of a rapid phase-shift reversal on a Jamaican north coast coral reef after the 2005 bleaching event. Revista de biología tropical 56(Suppl1):149-159.

Rambahiniarison, J. M., and coauthors. 2018. Life history, growth, and reproductive biology of four mobulid species in the Bohol Sea, Philippines. Frontiers in Marine Science 5:269.

Reddering, J. S. V. 1988. Prediction of the effects of reduced river discharge on estuaries of the south-eastern Cape Province, South Africa. South African Journal of Science 84:726-730.

Redfoot, W., and L. Ehrhart. 2013. Trends in size class distribution, recaptures, and abundance of juvenile green turtles (*Chelonia mydas*) utilizing a rock riprap lined embayment at Port Canaveral, Florida, USA, as developmental habitat. Chelonian Conservation and Biology 12(2):252–261.

Reece, J. S., T. A. Castoe, and C. L. Parkinson. 2005. Historical perspectives on population genetics and conservation of three marine turtle species. Conservation Genetics 6(2):235-251.

Reine, K. J., and coauthors. 2014. Assessing impacts of navigation dredging on Atlantic sturgeon (*Acipenser oxyrinchus*): Final report. U.S. Department of Defense, Army Corps of Engineers, Engineer Research and Development Center, ERDC/EL TR-14-12.

Reine, K. J., and C. Dickerson. 2014. Characterization of underwater sounds produced by a hydraulic dutterhead dredge during maintenance dredging in the Stockton Deepwater Shipping Channel, California. U.S. Army Corps of Engindeers, Engineer Research and Development Center, ERDC TN-DOER-E38, Vicksburg, MS.

Reiser, C. M., D. W. Funk, R. Rodrigues, and D. Hannay. 2010. Marine mammal monitoring and mitigation during marine geophysical surveys by Shell Offshore, Inc. in the Alaskan Chukchi and Beaufort seas, July-October 2010: 90-day report. LGL Alaska Research Associates Inc. and JASCO Applied Sciences,, LGL Report P1171E-1, Anchorage, AK.

Rhodin, A. G. J. 1985. Comparative chondro-osseous development and growth in marine turtles. Copeia 1985(3):752-771.

Rice, K. J., and N. C. Emery. 2003. Managing microevolution: restoration in the face of global change. Frontiers in Ecology and the Environment 1(9):469-478.

Richards, P. M., and coauthors. 2011. Sea turtle population estimates incorporating uncertainty: A new approach applied to western North Atlantic loggerheads *Caretta caretta*. Endangered Species Research 15(2):151-158.

Richardson, B., and D. H. Secor. 2017. Assess threats to the reproduction by Atlantic sturgeon through studies on spawning habitats of Chesapeake Bay DPS sturgeon in the Nanticoke estuary. Maryland Department of Natural Resources.

Richardson, W. J., and coauthors. 1995. Acoustic effects of oil production activities on bowhead and white whales visible during spring migration near Pt. Barrow, Alaska--1991 and 1994 phases: Sound propagation and whale responses to playbacks of icebreaker noise. LGL Ltd., environmental research associates and Greeneridge Sciences Inc., MMS 95-0051, Herndon, VA.

Richmond, R. H. 1993. Coral reefs: Present problems and future concerns resulting from anthropogenic disturbance. American Zoologist 33(6):524-536.

Richter, C., S. Dawson, and E. Slooten. 2006. Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 22(1):46-63.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin, and J. H. Anderson. 1969. Hearing in the fiant sea turtle, *Chelonia mydas*. Proceedings of the National Academy of Sciences 64(3):884-890.

Riegl, B. 1995. A revision of the hard coral genus *Acropora* Oken, 1815 (Scleractinia, Astrocoeniina, Acroporidae) in south-east Africa. Zoological Journal of the Linnean Society 113(3):249-288.

Riegl, B., and G. M. Branch. 1995. Effects of sediment on the energy budgets of four scleractinian (Bourne 1900) and five alcyonacean (Lamouroux 1816) corals. Journal of Experimental Marine Biology and Ecology 186(2):259-275.

Riegl, B., S. J. Purkis, J. Keck, and G. P. Rowlands. 2009. Monitored and modeled coral population dynamics and the refuge concept. Marine Pollution Bulletin 58(1):24-38.

Ritson-Williams, R., V. J. Paul, S. N. Arnold, and R. S. Steneck. 2010. Larval settlement preferences and post-settlement survival of the threatened Caribbean corals *Acropora palmata* and *A. cervicornis*. Coral Reefs 29(1):71-81.

Rivalan, P., and coauthors. 2005. Trade-off between current reproductive effort and delay to next reproduction in the leatherback sea turtle. Oecologia 145(4):564-574.

Roberts, J. J., and coauthors. 2016. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. Scientific Reports 6(1):22615.

Robinson, R., and coauthors. 2005. Climate change and migratory species. British Trust for Ornithology Research Report 414, Defra Research Contract CR0302.

Rogers, C. S. 1983. Sublethal and lethal effects of sediments applied to common Caribbean reef corals in the field. Marine Pollution Bulletin 14(10):378-382.

Rogers, C. S. 1990. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62(1):185-202.

Rogers, C. S., and E. M. Muller. 2012. Bleaching, disease and recovery in the threatened scleractinian coral *Acropora palmata* in St. John, US Virgin Islands: 2003–2010. Coral Reefs 31(3):807-819.

Rogers, C. S., T. H. Suchanek, and F. A. Pecora. 1982. Effects of Hurricanes David and Frederic (1979) on shallow *Acropora palmata* reef communities: St. Croix, U.S. Virgin Islands. Bulletin of Marine Science 32(2):532-548.

Rogers, S. G., and W. Weber. 1994. Occurrence of shortnose sturgeon (Acipenser brevirostrum) in the Ogeechee-Canoochee river system, Georgia, during the summer of 1993.

Rogers, S. G., and W. Weber. 1995. Status and restoration of Atlantic and shortnose sturgeons in Georgia. Georgia Department of Natural Resources, Coastal Resources Divsion, Anadromous Grants Program Project Award Number NA46FA102-01, Brunswick, GA.

Rolland, R. M., and coauthors. 2012. Evidence that ship noise increases stress in right whales. Proceedings of the Royal Society B: Biological Sciences 279(1737):2363-2368.

Romanov, E. V. 2002. Bycatch in the tuna purse-seine fisheries of the western Indian Ocean. Fishery Bulletin 100(1):90-105.

Ross, D. 1976. Mechanics of Underwater Noise. Pergamon Press, New York.

Ross, D. 1993. On ocean underwater ambient noise. Acoustics Bulletin 18:5-8.

Ross, D. 2005. Ship sources of ambient noise. IEEE Journal of Oceanic Engineering 30(2):257-261.

Rubin, R. D., K. R. Kumli, and G. Chilcott. 2008. Dive characteristics and movement patterns of acoustic and satellite-tagged manta rays (*Manta birostris*) in the Revillagigedos Islands of Mexico. American Elasmobranch Society, Montreal, Canada.

Ruddle, V. K. 2018. Age Structure, Reproduction, and Recruitment of Atlantic sturgeon (Acipenser oxyrinchus) and Shortnose sturgeon (Acipenser brevirostrum) in the Cooper River, South Carolina. College of Charlesston.

Rudloe, J. 1981. From the Jaws of Death. Pages 60-70 *in*. Sports Illustrated.

Ruelle, R., and C. Henry. 1992. Organochlorine compounds in pallid sturgeon. U.S. Department of the Interior, U.S. Fish and Wildlife Service, South Dakota State Office, Pierre, SD.

Ruelle, R., and K. D. Keenlyne. 1993. Contaminants in Missouri River pallid sturgeon. Bulletin of Environmental Contamination and Toxicology 50(6):898-906.

Ruhl, J. B. 2003. Equitable apportionment of ecosystem service: New water law for a new water age. Journal of Land Use & Environmental Law 19(1):47-57.

Saba, V. S., and coauthors. 2016. Enhanced warming of the Northwest Atlantic Ocean under climate change. Journal of Geophysical Research: Oceans 121(1):118-132.

Saeki, K., H. Sakakibara, H. Sakai, T. Kunito, and S. Tanabe. 2000. Arsenic accumulation in three species of sea turtles. BioMetals 13(3):241-250.

SAFMC. 1998. Final habitat plan for the South Atlantic region: Essential fish habitat requirements for fishery management plans of the South Atlantic Fishery Management Council. South Atlantic Fishery Management Council, Charleston, South Carolina.

Sahoo, G., R. K. Sahoo, and P. Mohanty-Hejmadi. 1996. Distribution of heavy metals in the eggs and hatchlings of olive ridley sea turtles, *Lepidochelys olivacea*, from Gahirmatha, Orissa. Indian Journal of Marine Sciences 25(4):371-372.

Salwasser, H., S. E. Mealey, and K. Johnson. 1984. Wildlife population viability: A question of risk, volume 49. Wildlife Managment Institute, Washington, D.C.

Santidrián Tomillo, P., and coauthors. 2007. Reassessment of the leatherback turtle (*Dermochelys coriacea*) nesting population at Parque Nacional Marino Las Baulas, Costa Rica: Effects of conservation efforts. Chelonian Conservation and Biology 6(1):54-62.

Sargent, F. J., T. J. Leary, D. W. Crewz, and C. R. Kruer. 1995. Scarring of Florida's seagrasses: Assessment and management options. Florida Marine Research Institute, FMRI Technical Report TR-1, Saint Petersburg, FL.

Savoy, T. 2007. Prey eaten by Atlantic sturgeon in Connecticut waters. American Fisheries Society Symposium 56:157-165.

Savoy, T., L. Maceda, N. K. Roy, D. Peterson, and I. Wirgin. 2017. Evidence of natural reproduction of Atlantic sturgeon in the Connecticut River from unlikely sources. PLOS ONE 12(4):e0175085-e0175085.

Savoy, T., and D. Pacileo. 2003. Movements and important habitats of subadult Atlantic sturgeon in Connecticut waters. Transactions of the American Fisheries Society 132(1):1-8.

SBNMS. 2009. Right whale strike statement involving Stellwagen Bank National Marine Sanctuary's RV AUK, April 19, 2009.

Schärer, M., and coauthors. 2009. Elkhorn coral distribution and condition throughout the Puerto Rican archipelago, Ft. Lauderdale, FL.

Scharer, R. M., W. F. Patterson III, J. K. Carlson, and G. R. Poulakis. 2012. Age and growth of endangered smalltooth sawfish (*Pristis pectinata*) verified with LA-ICP-MS analysis of vertebrae. PLOS ONE 7:e47850.

Schelten, C., S. Brown, C. B. Gurbisz, B. Kautz, and J. A. Lentz. 2006. Status of *Acropora palmata* populations off the coast of South Caicos, Turks and Caicos Islands. Gulf and Caribbean Fisheries Institute, Sanit Petersburg, FL.

Schmid, J. R., and W. J. Barichivich. 2006. *Lepidochelys kempii* – Kemp's ridley. Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3:128-141.

Schmid, J. R., and A. Woodhead. 2000. Von Bertalanffy growth models for wild Kemp's ridley turtles: Analysis of the NMFS Miami Laboratory tagging database. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-444, Miami, FL.

Scholz, N. L., and coauthors. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 57(9):1911-1918.

Schopmeyer, S. A., and coauthors. 2012. In situ coral nurseries serve as genetic repositories for coral reef restoration after an extreme cold-water event. Restoration Ecology 20(6):696-703.

Schopmeyer, S. A., and coauthors. 2017. Regional restoration benchmarks for *Acropora cervicornis*. Coral Reefs 36(4):1047-1057.

Schroeder, B. A., and A. M. Foley. 1995. Population studies of marine turtles in Florida Bay. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-361, Miami, FL.

Schueller, P., and D. L. Peterson. 2010. Abundance and recruitment of juvenile Atlantic sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 139(5):1526-1535.

Schuhmacher, H., and H. Zibrowius. 1985. What is hermatypic? A redefinition of ecological groups in corals and other organisms. Coral Reefs 4(1):1-9.

Schulz, J. P. 1975. Sea turtles nesting in Surinam. Zoologische Verhandelingen 143:141.

Schwartz, F. J. 1978. Behavioral and tolerance responses to cold water temperatures by three species of sea turtles (Reptilia, Cheloniidae) in North Carolina. Florida Marine Research Publications 33:16-18.

Scott, W. B., and E. J. Crossman. 1973. Freshwater fishes of Canada., Fisheries Research Board of Canada Bulletin.

Secor, D. H. 1995. Chesapeake Bay Atlantic sturgeon: Current status and future recovery. University of Maryland, Center for Estuarine and Environmental Studies, Chesapeake Bay Biological Laboratory, Solomons, MD.

Secor, D. H. 2002. Atlantic sturgeon fisheries and stock abundances during the late nineteenth century. Pages 89-100 *in* W. van Winkle, P. J. Anders, D. H. Secor, and D. A. Dixon, editors. Biology, Management, and Protection of North American Sturgeon, volume 28. American Fisheries Society, Saint Louis, MO.

Secor, D. H., and T. E. Gunderson. 1998. Effects of hypoxia and temperature on survival, growth, and respiration of juvenile Atlantic sturgeon (*Acipenser oxyrinchus*). Fishery Bulletin 96:603-613.

Secor, D. H., and E. J. Niklitschek. 2001. Hypoxia and sturgeons. Report to the Chesapeake Bay Program Dissolved Oxygen Criteria Team. University of Maryland Center for

Environmental Science, Chesapeake Biological Laboratory, UMCES Techincal Series No. TS-314-01-CBL, Reference No. [UMCES] CBL 01-0080, Solomons, MD.

Secor, D. H., and coauthors. 2000. Dispersal and growth of yearling Atlantic sturgeon, *Acipenser oxyrinchus*, released into Chesapeake Bay. Fishery Bulletin 98(4):800-810.

Secor, D. H., and J. R. Waldman. 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. Pages 203-216 *in* J. A. Musick, editor. Life in the Slow Lane: Ecology and Conservation of Long-Lived Marine Animals, volume 23. American Fisheries Society, Bethesda, MD.

Seitz, J. C., and G. R. Poulakis. 2002. Recent occurrence of sawfishes (Elasmobranchiomorphi: Pristidae) along the southwest coast of Florida (USA). Florida Scientist 65(4):256-266.

Seminoff, J. A., and coauthors. 2015. Status review of the green turtle (*Chelonia mydas*) under the Endangered Species Act. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, NOAA Technical Memorandum NMFS-SWFSC-539, La Jolla, CA.

Shaffer, M. L. 1981. Minimum population sizes for species conservation. BioScience 31(2):131-134.

Shaver, D. J. 1994. Relative abundance, temporal patterns, and growth of sea turtles at the Mansfield Channel, Texas. Journal of Herpetology 28(4):491-497.

Shaver, D. J. 2002. Green sea turtles (*Chelonia mydas*) in a south Texas (USA) developmental habitat. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-477, Miami, FL.

Shenker, J. M. 1984. Scyphomedusae in surface waters near the Oregon coast, May-August, 1981. Estuarine, Coastal and Shelf Science 19(6):619-632.

Sheppard, C., and R. Rioja-Nieto. 2005. Sea surface temperature 1871–2099 in 38 cells in the Caribbean region. Marine Environmental Research 60(3):389-396.

Sherk Jr., J. A. 1972. Current status of the knowledge of the biological effects of suspended and deposited sediments in Chesapeake Bay. Chesapeake Science 13:S137-S144.

Shillinger, G. L., and coauthors. 2008. Persistent leatherback turtle migrations present opportunities for conservation. PLoS Biology 6(7):1408-1416.

Shinn, E. A. 1963. Spur and groove formation on the Florida Reef Tract. Journal of Sedimentary Petrology 33(2):291-303.

Shoop, C. R., and R. D. Kenney. 1992. Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. Herpetological Monographs 6:43-67.

Short, F. T., T. J. B. Carruthers, W. Dennison, and M. Waycott. 2007. Global seagrass distribution and diversity: A bioregional model. Journal of Experimental Marine Biology and Ecology 350(1-2):3-20.

Short, F. T., and H. A. Neckles. 1999. The effects of global climate change on seagrasses. Aquatic Botany 63(3):169-196.

Simmonds, M. P., and S. J. Isaac. 2007. The impacts of climate change on marine mammals: Early signs of significant problems. Oryx 41(1):19-26.

Simpfendorfer, C., G. Poulakis, P. M. O'Donnell, and T. R. Wiley. 2008. Growth rates of juvenile smalltooth sawfish *Pristis pectinata* Latham in the Western Atlantic. Journal of Fish Biology 72:711-723.

Simpfendorfer, C. A. 2000. Predicting population recovery rates for endangered western Atlantic sawfishes using demographic analysis. Environmental Biology of Fishes 58(4):371-377.

Simpfendorfer, C. A. 2001. Essential habitat of the smalltooth sawfish, *Pristis pectinata*. Mote Marine Laboratory, Center for Shark Research, Technical Report 786, Sarasota, FL.

Simpfendorfer, C. A. 2002. Smalltooth sawfish: The USA's first endangered elasmobranch? Endangered Species Update 19(3):53-57.

Simpfendorfer, C. A. 2003. Abundance, movement and habitat use of the smalltooth sawfish. Final Report. Mote Marine Laboratory Mote Technical Report No. 929, Sarasota, FL.

Simpfendorfer, C. A. 2006. Movement and habitat use of smalltooth sawfish. Final Report. Mote Marine Laboratory, Mote Marine Laboratory Technical Report 1070, Sarasota, FL.

Simpfendorfer, C. A., and T. R. Wiley. 2004. Determination of the distribution of Florida's remnant sawfish population and identification of areas critical to their conservation. Mote Marine Laboratory, Technical Report NG-01-001, Sarasota, FL.

Simpfendorfer, C. A., and T. R. Wiley. 2005. Identification of priority areas for smalltooth sawfish conservation. Final report to the National Fish and Wildlife Foundation for Grant # 2003-0041-000. Mote Marine Laboratory.

Simpfendorfer, C. A., T. R. Wiley, and B. G. Yeiser. 2010. Improving conservation planning for an endangered sawfish using data from acoustic telemetry. Biological Conservation 143(6):1460-1469.

Simpfendorfer, C. A., and coauthors. 2011. Environmental influences on the spatial ecology of juvenile smalltooth sawfish (*Pristis pectinata*): results from acoustic monitoring. PLOS ONE 6(2):e16918.

Sindermann, C. J. 1994. Quantitative effects of pollution on marine and anadromous fish populations. U.S. Department of Commerce, National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, NOAA Technical Memorandum NMFS-F/NEC-104, Woods Hole, MA.

Smith, J. A., H. J. Flowers, and J. E. Hightower. 2014. Fall spawning of Atlantic sturgeon in the Roanoke River, North Carolina. Transactions of the American Fisheries Society 144(1):48-54.

Smith, T. B. 2013. United States Virgin Islands' response to the proposed listing or change in status of seven Caribbean coral species under the U.S. Endangered Species Act University of the Virgin Islands, Center for Marine and Environmental Studies.

Smith, T. B., and coauthors. 2011. The United States Virgin Islands Territorial Coral Reef Monitoring Program. 2011 Annual Report. Version 1. The Center for Marine and Environmental Studies, University of the Virgin Islands, The Division of Coastal Zone Management, USVI Department of Planning and Natural Resources, and The Coral Reef Conservation Program, National Oceanic and Atmospheric Administration.

Smith, T. I. J. 1985. The fishery, biology, and management of Atlantic sturgeon, *Acipenser oxyrhynchus*, in North America. Environmental Biology of Fishes 14(1):61-72.

Smith, T. I. J., and J. P. Clugston. 1997. Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. Environmental Biology of Fishes 48(1-4):335-346.

Smith, T. I. J., E. K. Dingley, and D. E. Marchette. 1980. Induced spawning and culture of Atlantic sturgeon. The Progressive Fish-Culturist 42(3):147-151.

Smith, T. I. J., D. E. Marchette, and R. A. Smiley. 1982. Life history, ecology, culture and management of Atlantic sturgeon, *Acipenser oxyrhynchus*, Mitchill, in South Carolina. South Carolina Wildlife and Marine Resources, Resources Department, Final Report to U.S. Fish and Wildlife Service, Project AFS-9.

Smultea, M. A., J. L. Hopkins, and A. M. Zoidis. 2008a. Marine mammal and sea turtle monitoring survey in support of Navy training exercises in the Hawai'i Range Complex November 11-17, 2007. Department of the Navy, NAVFAC Pacific, Oakland, California.

Smultea, M. A., J. R. Mobley Jr., D. Fertl, and G. L. Fulling. 2008b. An unusual reaction and other observations of sperm whales near fixed-wing aircraft. Gulf and Caribbean Research 20:75-80.

Snelson, F., and S. Williams. 1981. Notes on the occurrence, distribution, and biology of *elasmobranch* fishes in the Indian River lagoon system, Florida. Estuaries and Coasts 4(2):110-120.

Snoddy, J. E., M. Landon, G. Blanvillain, and A. Southwood. 2009. Blood biochemistry of sea turtles captured in gillnets in the Lower Cape Fear River, North Carolina, USA. Journal of Wildlife Management 73(8):1394-1401.

Snoddy, J. E., and A. S. Williard. 2010. Movements and post-release mortality of juvenile sea turtles released from gillnets in the lower Cape Fear River, North Carolina, USA. Endangered Species Research 12(3):235-247.

Snover, M. L. 2002. Growth and ontogeny of sea turtles using skeletochronology: Methods, validation and application to conservation. Dissertation. Duke University, Durham, NC.

Soong, K., and J. C. Lang. 1992. Reproductive integration in reef corals. Biological Bulletin 183(3):418-431.

Soulé, M. E. 1980. Thresholds for survival: Maintaining fitness and evolutionary potential. Pages 151-1769 *in* M. E. Soulé, and B. A. Wilcox, editors. Conservation Biology: An Evolutionary-Ecological Perspective. Sinauer Associates, Sunderland, MA.

Soulé, M. E. 1987. Where do we go from here? Chapter 10 In: Soulé, M.E. (ed), Viable Populations for Conservation. Cambridge University Press, pp.175-183.

Southall, B. L., and coauthors. 2007. Marine mammal noise exposure criteria: Initial scientific recommendations. Aquatic Mammals 33(4):411-521.

Southwood, A. L., R. D. Andrews, F. V. Paladino, and D. R. Jones. 2005. Effects of diving and swimming behavior on body temperatures of Pacific leatherback turtles in tropical seas. Physiological and Biochemical Zoology 78(2):285-297.

Spotila, J. R., and coauthors. 1996. Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? Chelonian Conservation and Biology 2(2):209-222.

Spotila, J. R., R. D. Reina, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. 2000. Pacific leatherback turtles face extinction. Nature 405:529-530.

Squiers, T. S. 2004. State of Maine 2004 Atlantic sturgeon compliance report to the Atlantic States Marine Fisheries Commission, Washington, D.C.

Stabenau, E. K., T. A. Heming, and J. F. Mitchell. 1991. Respiratory, acid-base and ionic status of Kemp's ridley sea turtles (*Lepidochelys kempii*) subjected to trawling. Comparative Biochemistry and Physiology Part A: Physiology 99A(1/2):107-111.

Stabenau, E. K., and K. R. N. Vietti. 2003. The physiological effects of multiple forced submergences in loggerhead sea turtles (*Caretta caretta*). Fishery Bulletin 101(4):889-899.

Stacy, B. A., J. L. Keene, and B. A. Schroeder. 2016. Report of the technical expert workshop: Developing national criteria for assessing post-interaction mortality of sea turtles in trawl, net, and pot/trap fisheries. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, NOAA Technical Memorandum NMFS-OPR-53, Silver Sring, MD.

Starbird, C. H., A. Baldridge, and J. T. Harvey. 1993. Seasonal occurrence of leatherback sea turtles (*Dermochelys coriacea*) in the Monterey Bay region, with notes on other sea turtles, 1986-1991. California Fish and Game 79(2):54-62.

Starbird, C. H., and M. M. Suarez. 1994. Leatherback sea turtle nesting on the north Vogelkop coast of Irian Jaya and the discovery of a leatherback sea turtle fishery on Kei Kecil Island. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memorandum NMFS-SEFSC-351, Miami, FL.

Steadman, S., and T. E. Dahl. 2008. Status and trends of wetlands in the coastal watersheds of the Eastern United States 1998 to 2004. Pages 32 *in* N. M. F. S. a. U. S. D. o. t. National Oceanic and Atmospheric Administration, and F. a. W. S. Interior, editors.

Stein, A. B., K. D. Friedland, and M. R. Sutherland. 2004a. Atlantic sturgeon marine bycatch and mortality on the continental shelf of the Northeast United States. North American Journal of Fisheries Management 24(1):171-183.

Stein, A. B., K. D. Friedland, and M. R. Sutherland. 2004b. Atlantic sturgeon marine distribution and habitat use along the northeastern coast of the United States. Transactions of the American Fisheries Society 133:527-537.

Steiner, S. C. C. 2003. Stony corals and reefs of Dominica. Atoll Research Bulletin 498(497):1-15.

Stephenson, W., S. D. Cook, and S. J. Newlands. 1978. The macrobenthos of the Middle Banks area of Moreton Bay. Memoirs of the Queensland Museum 18(2):95-118.

Stevenson, J. T., and D. H. Secor. 1999. Age determination and growth of Hudson River Atlantic sturgeon (*Acipenser oxyrinchus*). Fishery Bulletin 97(1):153-166.

Steward, J. S., and coauthors. 2006. The impacts of the 2004 hurricanes on hydrology, water quality, and seagrass in the central Indian River Lagoon, Florida. Estuaries and Coasts 29(6):954-965.

Stewart, J. D., E. M. Hoyos-Padilla, K. R. Kumli, and R. D. Rubin. 2016. Deep-water feeding and behavioral plasticity in *Manta birostris* revealed by archival tags and submersible observations. Zoology 119.

Stewart, J. D., M. Nuttall, E. L. Hickerson, and M. A. Johnston. 2018. Important juvenile manta ray habitat at Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. Marine Biology 165:111.

Stewart, K. R., and C. Johnson. 2006. *Dermochelys coriacea* - Leatherback sea turtle. Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3:144-157.

Stewart, K. R., C. Johnson, and M. H. Godfrey. 2007. The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. Herpetological Journal 17(2):123-128.

Stewart, K. R., J. M. Keller, R. Templeton, J. R. Kucklick, and C. Johnson. 2011. Monitoring persistent organic pollutants in leatherback turtles (*Dermochelys coriacea*) confirms maternal transfer. Marine Pollution Bulletin 62(7):1396-1409.

Steyermark, A. C., and coauthors. 1996. Nesting leatherback turtles at Las Baulas National Park, Costa Rica. Chelonian Conservation and Biology 2(2):173-183.

Stickney, R. R., and D. Perlmutter. 1975. Impact of Intracoastal Waterway maintenance dredging on a mud bottom benthos community. Biological Conservation 7(3):211-225.

Storelli, M. M., G. Barone, and G. O. Marcotrigiano. 2007. Polychlorinated biphenyls and other chlorinated organic contaminants in the tissues of Mediterranean loggerhead turtle *Caretta caretta*. Science of the Total Environment 373(2-3):456-463.

Storelli, M. M., G. Barone, A. Storelli, and G. O. Marcotrigiano. 2008. Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. Chemosphere 70(5):908-913.

Strayer, D. L. 2010. Alien species in fresh waters: Ecological effects, interactions with other stressors, and prospects for the future. Freshwater Biology 55:152-174.

Suchman, C. L., and R. D. Brodeur. 2005. Abundance and distribution of large medusae in surface waters of the northern California Current. Deep Sea Research Part II: Topical Studies in Oceanography 52(1–2):51-72.

Sweka, J. A., and coauthors. 2007. Juvenile Atlantic sturgeon habitat use in Newburgh and Haverstraw Bays of the Hudson River: Implications for population monitoring. North American Journal of Fisheries Management 27(4):1058-1067.

Swingle, W. M., S. G. Barco, T. D. Pitchford, W. A. McLellan, and D. A. Pabst. 1993. Apperance of juvenile humpback whales feeding in the nearshore waters of Virginia. Marine Mammal Science 9(3):309-315.

Szmant, A. M. 1986. Reproductive ecology of Caribbean reef corals. Coral Reefs 5(1):43-53.

Szmant, A. M., and M. W. Miller. 2006. Settlement preferences and post-settlement mortality of laboratory cultured and settled larvae of the Caribbean hermatypic corals *Montastaea faveolata* and *Acropora palmata* in the Florida Keys, U.S.A., Okinawa, Japan.

Szmant, A. M., E. Weil, M. W. Miller, and D. E. Colón. 1997. Hybridization within the species complex of the scleractinan coral *Montastraea annularis*. Marine Biology 129(4):561-572.

Taubert, B. D. 1980. Biology of the shortnose sturgeon (*Acipenser brevirostrum*) in Holyoke Pool, Connecticut River, Massachusetts. University of Massachusetts.

Thomas, C. D. 1990. What do real population dynamics tell us about minimum viable population sizes? Conservation Biology 4(3):324-327.

Thomsen, F., S. R. McCully, D. Wood, F. Pace, and P. White. 2009. A generic investigation into noise profiles of marine dredging in relation to the acoustic sensitivity of the marine fauna in UK waters: PHASE 1 Scoping and review of key issues, MEPF Ref No. MEPF/08/P21.

Tiwari, M., B. P. Wallace, and M. Girondot. 2013. *Dermochelys coriacea* (Northwest Atlantic Ocean subpopulation), Leatherback. The IUCN Red List of Threatened Species, International Union for Conservation of Nature and Natural Resources.

Tomascik, T. 1990. Growth rates of two morphotypes of *Montastrea annularis* along a eutrophication gradient, Barbados, W.I. Marine Pollution Bulletin 21(8):376-381.

Tomascik, T., and F. Sander. 1987. Effects of eutrophication on reef-building corals. II. Structure of scleractinian coral communities on fringing reefs, Barbados, West Indies. Marine Biology 94(1):53-75.

Torquemada, Y. F., M. J. Durako, and J. L. S. Lizaso. 2005. Effects of salinity and possible interactions with temperature and pH on growth and photosynthesis of *Halophila johnsonii* Eiseman. Marine Biology 148(2):251-260.

Troëng, S. 1998. Poaching threatens the green turtle rookery at Tortuguero, Costa Rica. Marine Turtle Newsletter 79:11-12.

Troëng, S., D. Chacón, and B. Dick. 2004. Possible decline in leatherback turtle *Dermochelys coriacea* nesting along the coast of Caribbean Central America. Oryx 38(4):395-403.

Troëng, S., E. Harrison, D. Evans, A. de Haro, and E. Vargas. 2007. Leatherback turtle nesting trends and threats at Tortuguero, Costa Rica. Chelonian Conservation and Biology 6(1):117-122.

Troëng, S., and E. Rankin. 2005. Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biological Conservation 121:111-116.

Tsyplakov, E. P. 1978. Migrations and distribution of the starlet, Acipenser ruthenus, in Kuybyshev reservoir Journal of Ichthyology 18:905-912.

Tucker, A. D. 1988. A summary of leatherback turtle *Dermochelys coriacea* nesting at Culebra, Puerto Rico from 1984-1987 with management recommendations. U.S. Department of the Interior, Fish and Wildlife Service, Athens GA.

Tucker, A. D. 2010. Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: Implications for stock estimation. Journal of Experimental Marine Biology and Ecology 383(1):48-55.

Tunnicliffe, V. 1981. Breakage and propagation of the stony coral *Acropora cervicornis*. Proceedings of the National Academy of Sciences 78(4):2427-2431.

Turtle Expert Working Group. 1998. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the western North Atlantic. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-409, Miami, FL.

Turtle Expert Working Group. 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-444, Miami, FL.

Turtle Expert Working Group. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-555, Miami, FL.

Turtle Expert Working Group. 2009. An assessment of the loggerhead turtle population in the western North Atlantic ocean. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-575, Miami, FL.

U.S. Global Change Research Program. 2001. Climate change impacts on the United States: The potential consequences of climate variability and change. U.S. Global Change Research Program, National Assessment Synthesis Team, Washington, D.C.

U.S. Global Change Research Program. 2018. Fourth national climate assessment. Impacts, risks, and adaptation in the United States. Volume II. U.S. Global Change Research Program, Washington, D.C.

U.S. Navy. 2017. Criteria and thresholds for U.S. Navy acoustic and explosive effects analysis (Phase III). U.S. Department of Defense, U.S. Department of the Navy, Technical Report.

USACE. 1993. Dredging Fundamentals Facilitator's Guide. U.S. Department of Defense, Army Corps of Engineers, Huntsville Division.

USACE. 2008. Engineering and Design Coastal Engineering Manual, EM 1110-2-1100.

USACE. 2013. Jacksonville Harbor Navigation Study, Duval County, Florida: Draft integrated general re-evaluation report II and supplemental environmental impact statement. U.S.

Department of Defense, U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL.

USACE. 2015a. Bed leveler evaluation report. U.S. Department of Defense, Army Corps of Engineers, Savannah District.

USACE. 2015b. Dredging and dredged material management. U.S. Department of Defense, U.S. Army Corps of Engineers, EM 1110-2-5025, Washington, D.C.

USACE. 2017. South Atlantic Regional Biological Assessment (SARBA). U.S. Department of Defense, U.S. Army Coprs of Engineers, South Atlantic Divsion and U.S. Department of the Interior, Bureau of Ocean Energy Management.

USACE. 2019a. Evaluating effects of dredging-induced underwater sound on aquatic species: A literature review. U.S. Department of Defense, Army Corps of Engineers, Engineer Research and Development Center, Dredging Operations and Environmental Research (DOER), ERDC/EL TR-19-18, Vicksburg, MS

USACE. 2019b. Savannah River Harbor cutterhead dredging dissolved oxygen draft project report: Monitoring the effects of hydraulic-cutterhead dredging on dissolved oxygen in the Savannah River Harbor. U.S. Department of Defense, Army Corps of Engineers, Engineer Research and Development Center.

USACE, and USEPA. 2008. Requirements and Procedures for Evaluation of the Ocean Disposal of Dredged Material in Southeastern U.S. Atlantic and Gulf Coast Waters Pages 447 *in*, Atlanta GA.

USEPA. 2012. Climate Change Adaptation Plan. U.S. Environmental Protection Agency.

USFWS. 1993. Pallid sturgeon (*Scaphirhynchus albus*) recovery plan. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Bismarck, ND.

USFWS. 2003. Draft Fish and Wildlife Coordination Act report on Savannah River Basin comprehensive study. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Southeast Region, Atlanta, GA.

USFWS and NMFS. 1998. Endangered Species Act consultation handbook. Procedures for Conducting Section 7 Consultations and Conferences. U.S. Fish and Wildlife, National Marine Fisheries Service.

USGRG. 2004. U.S. National Assessment of the Potential Consequences of Climate Variability and Change, Regional Paper: The Southeast. U.S. Global Research Group. Washington, D.C., August 20, 2004.

Vallarta, J., and R. McHugh. 2007. 3D passive acoustic localisation: Homogeneous vs non-homogeneous medium.

van der Graaf, I. C. J. 1987. The use of ploughs or bed-levelers in maintenance dredging. Pages 177- 195 *in* Maintenance Dredging. Thomas Telford, London, England.

van Dolah, R. F., D. R. Calder, and D. M. Knott. 1984. Effects of dredging and open water disposal in a South Carolina estuary. Estuaries 7(1):28-37.

van Dolah, R. F., D. R. Calder, D. M. Knott, and M. S. Maclin. 1979. Effects of dredging and unconfined disposal on macrobenthic communities in Sewee Bay, South Carolina. South Carolina Wildlife and Marine Resources Department, Marine Resources Center, Contract No. DACW-60-77-C-0013, Techincal Report 39, Charleston, SC.

van Eenennaam, J. P., and S. I. Doroshov. 1998. Effects of age and body size on gonadal development of Atlantic sturgeon. Journal of Fish Biology 53(3):624-637.

van Eenennaam, J. P., and coauthors. 1996. Reproductive conditions of the Atlantic sturgeon (*Acipenser oxyrinchus*) in the Hudson River. Estuaries 19(4):769-777.

Vardi, T. 2011. The threatened Atlantic elkhorn coral, *Acropora palmata*: Population dynamics and their policy implications. Dissertion. University of California, San Diego, San Diego, CA.

Vardi, T., D. E. Williams, and S. A. Sandin. 2012. Population dynamics of threatened elkhorn coral in the northern Florida Keys, USA. Endangered Species Research 19(2):157–169.

Vargas-Angel, B., S. B. Colley, S. M. Hoke, and J. D. Thomas. 2006. The reproductive seasonality and gametogenic cycle of *Acropora cervicornis* off Broward County, Florida, USA. Coral Reefs 25(1):110-122.

Vargas-Angel, B., J. D. Thomas, and S. M. Hoke. 2003. High-latitude *Acropora cervicornis* thickets off Fort Lauderdale, Florida, USA. Coral Reefs 22(4):465-473.

Venables, S. 2013. Short term behavioural responses of manta rays, *Manta alfredi*, to tourism interactions in Coral Bay, Western Australia. Thesis. Murdoch University.

Verna, T., and M. Pointon. 2000. Proceedings of the Western Dredging Association Twentieth Technical Conference and Thirty-Second Annual Texas A&M Dredging Seminar, June 25-28, 2000. Texas A & M University, Center For Dredging Studies, Warwick, RI.

Villinski, J. T. 2003. Depth-independent reproductive characteristics for the Caribbean reef-building coral *Montastraea faveolata*. Marine Biology 142(6):1043-1053.

Virnstein, R. W., and L. M. Hall. 2009. Northern range extension of the seagrasses *Halophila johnsonii* and *Halophila decipiens* along the east coast of Florida, USA. Aquatic Botany 90(1):89-92.

Virnstein, R. W., and L. J. Morris. 2007a. Distribution and abundance of *Halophila johnsonii* in the Indian River Lagoon: An update. St. Johns River Water Management District, Technical Memorandum No. 51, Palatka, FL.

Virnstein, R. W., and L. J. Morris. 2007b. Distribution and abundance of *Halophila johnsonii* in the Indian River Lagoon: An update. St. Johns River Water Management District, Palatka, Florida.

Virnstein, R. W., L. J. Morris, J. D. Miller, and R. Miller-Myers. 1997. Distribution and abundance of *Halophila johnsonii* in the Indian River Lagoon. St. Johns River Water Management District, Technical Memorandum No. 24, Palatka, FL.

Vollmer, S. V., and S. R. Palumbi. 2007. Restricted gene flow in the Caribbean staghorn coral *Acropora cervicornis*: Implications for the recovery of endangered reefs. Journal of Heredity 98(1):40-50.

von Westernhagen, H., and coauthors. 1981. Bioaccumulating substances and reproductive success in baltic flounder (*Platichthys flesus*). Aquatic Toxicology 1(2):85-99.

Waddell (editor), J. E. 2005. The state of coral reef ecosystems of the United States and Pacific freely associated states: 2005. U.S. Department of Commerce, National Oceanic and Atmospheric Administation, National Ocean Service, National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Assessment's Biogeography Team, NOAA Technical Memorandum NOS NCCOS 11, Silver Spring, MD.

Waddell, J. E., and A. M. Clarke (editors). 2008. The state of coral reef ecosystems of the United States and Pacific Freely Associated States: 2008. U.S. Department of Commerce, National Oceanic and Atmospheric Administation, National Ocean Service, National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Assessment's Biogeography Team, NOAA Technical Memorandum NOS NCCOS 73, Silver Spring, MD.

Waldman, J., and coauthors. 2018. Contemporary and historical effective population sizes of Atlantic sturgeon Acipenser oxyrinchus oxyrinchus. Conservation Genetics.

Waldman, J. R., C. Grunwald, J. Stabile, and I. I. Wirgin. 2002a. Impacts of life history and biogeography on the genetic stock structure of Atlantic sturgeon Acipenser oxyrinchus oxyrinchus, Gulf sturgeon A-oxyrinchus desotoi, and shortnose sturgeon A-brevirostrum. Journal of Applied Ichthyology 18(4-6):509-518.

Waldman, J. R., C. Grunwald, J. Stabile, and I. I. Wirgin. 2002b. Impacts of life history and biogeography on the genetic stock structure of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*, Gulf sturgeon *A. oxyrinchus desotoi*, and shortnose sturgeon *A. brevirostrum*. Journal of Applied Ichthyology 18(4-6):509-518.

Waldman, J. R., and I. I. Wirgin. 1998. Status and restoration options for Atlantic sturgeon in North America. Conservation Biology 12(3):631-638.

Walker, B. K., E. A. Larson, A. L. Moulding, and D. S. Gilliam. 2012. Small-scale mapping of indeterminate arborescent acroporid coral (*Acropora cervicornis*) patches. Coral Reefs 31(3):885-894.

Walker, T. A., and G. O'Donnell. 1981. Observations on nitrate, phosphate and silicate in Cleveland Bay, northern Queensland. Australian Journal of Marine and Freshwater Research 32(6):877-887.

Wallace, B. P., and coauthors. 2010. Global patterns of marine turtle bycatch. Conservation Letters:1-12.

Wallace, C. C. 1985. Reproduction, recruitment and fragmentation in nine sympatric species of the coral genus *Acropora*. Marine Biology 88(3):217-233.

Waring, C. P., and A. Moore. 2004. The effect of atrazine on Atlantic salmon (*Salmo salar*) smolts in fresh water and after sea water transfer. Aquatic Toxicology 66(1):93-104.

Watters, D. L., M. M. Yoklavich, M. S. Love, and D. M. Schroeder. 2010. Assessing marine debris in deep seafloor habitats off California. Mar Pollut Bull 60(1):131-8.

Way, F., C. Ahern, R. Semmes, and M. Goodrich. 2007. Effects of agitation dredging in Savannah Harbor.

Waycott, M., D. Freshwater, R. York, A. Calladine, and W. Kenworthy. 2002. Evolutionary trends in the seagrass genus Halophila (Thouars): Insights from molecular phylogeny. Bulletin of Marine Science 71:1299-1308.

Weber, W., and C. A. Jennings. 1996. Endangered species management plan for the shortnose sturgeon, *Acipenser brevirostrum*, Final report to Port Stewart Military Reservation, Fort Stewart, GA.

Weil, E., and N. Knowlton. 1994. A multi-character analysis of the Caribbean coral *Montastraea annularis* (Ellis and Solander, 1786) and its two sibling species, *M. faveolata* (Ellis and Solander, 1786) and *M. franksi* (Gregory, 1895). Bulletin of Marine Science 55(1):151-175.

Weishampel, J. F., D. A. Bagley, and L. M. Ehrhart. 2004. Earlier nesting by loggerhead sea turtles following sea surface warming. Global Change Biology 10:1424-1427.

Weishampel, J. F., D. A. Bagley, L. M. Ehrhart, and B. L. Rodenbeck. 2003. Spatiotemporal patterns of annual sea turtle nesting behaviors along an East Central Florida beach. Biological Conservation 110(2):295-303.

Welsh, S. A., S. M. Eyler, M. F. Mangold, and A. J. Spells. 2002. Capture locations and growth rates of Atlantic sturgeon in the Chesapeake Bay. American Fisheries Society Symposium 28:183-194.

Wershoven, J. L., and R. W. Wershoven. 1992. Juvenile green turtles in their nearshore habitat of Broward County, Florida: A five year review. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-302, Miami, FL.

Wheaton, J. L., and W. C. Jaap. 1988. Corals and other prominent benthic cnidaria of Looe Key National Marine Sanctuary, Florida. Florida Department of Natural REsources, Bureau of Marine Research, Publication No. 43, Saint Petersburg, FL.

Whitfield, A. K., and M. N. Bruton. 1989. Some biological implications of reduced freshwater inflow into eastern Cape estuaries: a preliminary assessment. South African Journal of Science 85:691-694.

Whitfield, P. E., W. J. Kenworthy, K. K. Hammerstrom, and M. S. Fonseca. 2002. The role of a hurricane in the expansion of disturbances initiated by motor vessels on seagrass banks. Journal of Coastal Research (37):86-99.

Wilber, D. H., and D. G. Clarke. 2007. Defining and assessing benthic recovery following dredging and dredged material disposal. Newman Printing Company, Bryan, TX.

Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States: Assessing the relative importance of habitat destruction, alien species, pollution, overexploitation, and disease. BioScience 48(8):607-615.

Wilcox, C., and coauthors. 2015. Understanding the sources and effects of abandoned, lost, and discarded fishing gear on marine turtles in northern Australia. Conservation Biology 29(1):198-206.

Wiley, D. N. 2016. Vessel strike mitigation lessons from direct observations involving two collisions between noncommercial vessels and North Atlantic right whales (*Eubalaena glacialis*). Marine Mammal Science 32(4):1501-1509.

Wiley, D. N., R. A. Asmutis, T. D. Pitchford, and D. P. Gannon. 1995. Stranding and mortality of humpback whales, *Megaptera novaeangliae*, in the mid-Atlantic and southeast United States, 1985-1992. Fishery Bulletin 93(1):196-205.

Wiley, T. R., and C. A. Simpfendorfer. 2007. Site fidelity/residency patterns/habitat modeling. Final Report to the National Marine Fisheries Service, Grant number WC133F-06-SE-2976. Mote Marine Laboratory.

Wilkens, J. L., A. W. Katzenmeyer, N. M. Hahn, J. J. Hoover, and B. C. Suedel. 2015. Laboratory test of suspended sediment effects on short-term survival and swimming performance of juvenile Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*, Mitchill, 1815). Journal of Applied Ichthyology 31(6):984-990.

Wilkinson, C. R. 2008. Status of Coral Reefs of the World: 2008. Australian Institute of Marine Science, Townsville, Australia.

Williams, D. E., and M. W. Miller. 2005. Coral disease outbreak: pattern, prevalence and transmission in *Acropora cervicornis*. Marine Ecology Progress Series 301:119-128.

Williams, D. E., and M. W. Miller. 2010. Stabilization of fragments to enhance asexual recruitment in *Acropora palmata*, a threatened Caribbean coral. Restoration Ecology 18(S2):446-451.

Williams, D. E., and M. W. Miller. 2012. Attributing mortality among drivers of population decline in *Acropora palmata* in the Florida Keys (USA). Coral Reefs 31(2):369-382.

Williams, D. E., M. W. Miller, A. J. Bright, R. E. Pausch, and A. Valdivia. 2017. Thermal stress exposure, bleaching response, and mortality in the threatened coral *Acropora palmata*. Marine Pollution Bulletin 124(1):189-197.

Williams, D. E., M. W. Miller, and K. L. Kramer. 2008. Recruitment failure in Florida Keys *Acropora palmata*, a threatened Caribbean coral. Coral Reefs 27(3):697-705.

Wilson, S. M., G. D. Raby, N. J. Burnett, S. G. Hinch, and S. J. Cooke. 2014. Looking beyond the mortality of bycatch: Sublethal effects of incidental capture on marine animals. Biological Conservation 171:61-72.

Winger, P. V., P. J. Lasier, D. H. White, and J. T. Seginak. 2000. Effects of contaminants in dredge material from the lower Savannah River. Archives of Environmental Contamination and Toxicology 38(1):128-136.

Wirgin, I. I., and coauthors. 2005. Range-wide population structure of shortnose sturgeon *Acipenser brevirostrum* based on sequence analysis of the mitochondrial DNA control region. Estuaries 28(3):406-421.

Wirgin, I. I., C. Grunwald, J. Stabile, and J. R. Waldman. 2007. Genetic evidence for relict Atlantic sturgeon stocks along the mid-Atlantic coast of the USA. North American Journal of Fisheries Management 27(4):1214-1229.

Wirgin, I. I., C. Grunwald, J. Stabile, and J. R. Waldman. 2010. Delineation of discrete population segments of shortnose sturgeon *Acipenser brevirostrum* based on mitochondrial DNA control region sequence analysis. Conservation Genetics 11(3):689-708.

Wirgin, I. I., and T. L. King. 2011. Mixed stock analysis of Atlantic sturgeon from coastal locales and a non-spawning river. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Region Sturgeon Workshop, Alexandria, VA.

Wirgin, I. I., L. Maceda, C. Grunwald, and T. L. King. 2015. Population origin of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* by-catch in U.S. Atlantic coast fisheries. Journal of Fish Biology 86(4):1251-1270.

Wirgin, I. I., and coauthors. 2000. Genetic structure of Atlantic sturgeon populations based on mitochondrial DNA control region sequences. Transactions of the American Fisheries Society 129(2):476-486.

Wirgin, I. I., J. R. Waldman, J. Stabile, B. A. Lubinski, and T. L. King. 2002. Comparison of mitochondrial DNA control region sequence and microsatellite DNA analyses in estimating population structure and gene flow rates in Atlantic sturgeon *Acipenser oxyrinchus*. Journal of Applied Ichthyology 18(4-6):313-319.

Witherington, B. E. 1992. Behavioral responses of nesting sea turtles to artificial lighting. Herpetologica 48(1):31-39.

Witherington, B. E. 2002. Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. Marine Biology 140(4):843-853.

Witherington, B. E., and K. A. Bjorndal. 1991. Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. Biological Conservation 55(2):139-149.

Witherington, B. E., M. Bresette, and R. M. Herren. 2006. *Chelonia mydas* - green turtle. Biology and Conservation of Florida Turtles. Chelonian Research Monographs No. 3 3:90-104.

Witherington, B. E., and L. M. Ehrhart. 1989a. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon system, Florida. Copeia (3):696-703.

Witherington, B. E., and L. M. Ehrhart. 1989b. Status and reproductive characteristics of green turtles (*Chelonia mydas*) nesting in Florida. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Center, NOAA Technical Memorandum NMFS-SEFSC-226, Panama City, FL.

Witherington, B. E., S. Hirama, and A. Moiser. 2003. Effects of beach armoring structures on marine turtle nesting: Final project report. Florida Fish and Wildlife Conservation Commission.

Witherington, B. E., S. Hirama, and A. Moiser. 2007. Changes to armoring and other barriers to sea turtle nesting following severe hurricanes striking Florida beaches: Final project report. Florida Fish and Wildlife Conservation Commission, Melbourne Beach, FL.

Witt, M. J., and coauthors. 2007. Prey landscapes help identify foraging habitats for leatherback turtles in the NE Atlantic. Marine Ecology Progress Series (337):231-243.

Witt, M. J., B. J. Godley, A. C. Broderick, R. Penrose, and C. S. Martin. 2006. Leatherback turtles, jellyfish and climate change in the northwest Atlantic: Current situation and possible future scenarios. Book of Abstracts. Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Witzell, W. N. 2002. Immature Atlantic loggerhead turtles (*Caretta caretta*): Suggested changes to the life history model. Herpetological Review 33(4):266-269.

Wrona, A., and coauthors. 2007. Restoring ecological flows to the lower Savannah River: A collaborative scientific approach to adaptive management. Pages 538-549 *in* T. C.

Rasmussen, G. D. Carroll, and A. P. Georgakakos, editors. Proceedings of the 2007 Georgia Water Resources Conference. University of Georgia.

Wursig, B., S. K. Lynn, T. A. Jeffereson, and K. D. Mullin. 1998. Behaviour of cetaceans in the northen Gulf of Mexico relative to survey ships and aircraft. Aquatic Mammals 24.1:41-50.

Young, J. R., T. B. Hoff, W. P. Dey, and J. G. Hoff. 1988. Management recommendations for a Hudson River Atlantic sturgeon fishery based on an age-structured population model. Pages 353-365 *in* C. L. Smith, editor. Fisheries Research in the Hudson River. State University of New York Press, Albany, NY.

Ziegeweid, J., C. Jennings, and D. Peterson. 2008. Thermal maxima for juvenile shortnose sturgeon acclimated to different temperatures. Environmental Biology of Fishes 82(3):299-307.

Zimmer, B., W. F. Precht, E. Hickerson, and J. Sinclair. 2006. Discovery of *Acropora palmata* at the Flower Garden Banks National Marine Sanctuary, northwestern Gulf of Mexico. Coral Reefs 25(2):192.

Zinno, F. R. 2012. Captura incidental de tortugas marinas en Bajos del Solís, Uruguay. Profundización en Ecología. Universidad de la República Uruguay, Montevideo, Uruguay.

Zubillaga, A. L., L. M. Marquez, A. Croquer, and C. Bastidas. 2008. Ecological and genetic data indicate recovery of the endangered coral *Acropora palmata* in Los Roques, Southern Caribbean. Coral Reefs 27(1):63-72.

Zug, G. R., and R. E. Glor. 1998. Estimates of age and growth in a population of green sea turtles (*Chelonia mydas*) from the Indian River Lagoon system, Florida: A skeletochronological analysis. Canadian Journal of Zoology 76(8):1497-1506.

Zug, G. R., and J. F. Parham. 1996. Age and growth in leatherback turtles, *Dermochelys coriacea*: A skeletochronological analysis. Chelonian Conservation and Biology 2(2):244-249.

Zurita, J. C., and coauthors. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL.

Zurita, J. C., B. Prezas, R. Herrera, and J. L. Miranda. 1994. Sea turtle tagging program in Quintana Roo, Mexico. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, NOAA Technical Memorandum NMFS-SEFSC-351, Miami, FL.

Zwinenberg, A. J. 1977. Kemp's ridley, *Lepidochelys kempii* (Garman, 1880), undoubtedly the most endangered marine turtle today (with notes on the current status of *Lepidochelys olivacea*). Bulletin of the Maryland Herpetological Society 13(3):170-192.

Zychov, M. 2013. Underwater Sound Modeling of Low Energy Geophysical Equipment Operations. JASCO Document 00600, Version 2.0. Technical report by JASCO Applied Sciences for CSA Ocean Sciences Inc.

Zychov, M., and J. MacDonnell. 2013. Sound Source Characterizations for the Collaborative Baseline Survey Offshore Massachusetts Final Report: Side Scan Sonar, Sub-Bottom Profiler, and the R/V Small Research Vessel experimental. JASCO Document 00413, Version 2.0. Technical report by JASCO Applied Sciences for the (US) Bureau of Ocean Energy Management.

Appendix A. South Atlantic Regional Biological Opinion (2020 SARBO) Project Design Criteria (PDCs) Overview and General PDCs

The 2020 SARBO PDCs are part of the proposed action and are nondiscretionary, meaning that a project authorized under 2020 SARBO must adhere to all applicable PDCs. PDCs minimizing the impact of take associated with the proposed action are also explicitly stated as a requirement of the RPMs/Terms and Conditions in Section 10.3 and 10.4. Projects must be designed to meet all PDC requirements based on project activity type, location, species or critical habitat in the project area, and equipment type(s) that will be used. However, not all PDCs will be applicable to all projects. For example, if a project does not include the use of relocation trawling, then compliance with relocation trawling PDCs would not be required.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not strictly adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Substantially Similar Projects or Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but are substantially similar, or projects with substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by National Marine Fisheries Service (NMFS) prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the 2020 SARBO, will require individual Section 7 consultation. In addition, any area previously authorized or permitted to be dredged or have material placed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

All PDCs were developed jointly with the U.S. Army Corps of Engineers (USACE) and Bureau of Ocean Energy Management (BOEM) and are designed to avoid or minimize impacts on Endangered Species Act (ESA)-listed species and critical habitat under NMFS purview where appropriate. Many of the terms used are standard to the industry or regulatory-specific (e.g., maintenance dredging, advanced maintenance). These terms are explained in the Project Description (Section 2 of the 2020 SARBO) and defined below for purposes of compliance with the PDCs.

The PDCs are divided as follows:

<u>General PDCs (Appendix B)</u>

A list of all of the general PDCs that apply to all projects. These are further divided as follows:

- Eligibility Criteria PDCs: PDCs that define if an activity meets the criteria to be covered under the 2020 SARBO (Appendix B Section 1)
- Standard/General PDCs: PDC requirements that apply to all projects covered under the 2020 SARBO (Appendix B Section 2)
- Equipment-Specific PDCs: PDCs that apply to specific equipment types covered under the 2020 SARBO (Appendix B Section 3). These include PDCs specific to hopper dredging (Section 3.1), munitions of explosive concern (MEC)/ unexploded ordinance (UXO)

screening (Section 3.2), cutterhead dredging (Section 3.3), bed-leveling (Section 3.4), and relocation trawling (Section 3.5).

Appendix C- Appendix H are topic specific PDCs that apply in addition to the PDCs in Appendix B as described below

- Appendix C. Coral PDCs
  PDCs that apply to projects occurring within the range of corals (as defined in this appendix).
- Appendix D. Johnson's seagrass PDCs
  PDCs that apply to projects occurring within the range of Johnson's seagrass (as defined in this appendix).
- Appendix E. Sturgeon PDCs
  PDCs that apply to projects occurring within rivers where sturgeon occur (as defined in this appendix).
- Appendix F. North Atlantic Right Whale Plan
  PDCs that apply to projects occurring within the range of North Atlantic right whales during the time they may be present (as defined in this appendix).
- Appendix G. Geophysical and Geotechnical (G&G) Survey PDCs
  PDCs that apply if a USACE project uses geophysical or geotechnical survey equipment.
- Appendix H. Protected Species Observer (PSO) PDCs
  PDCs that apply to work conducted by the Protected Species Observer handling ESA-listed species.
- Appendix I. Relocation trawling lazy line guidance. This is information provided to support PDC RELOCATE.3 in the General PDCs in Appendix B.
- Appendix J. Risk assessment guidance. This guidance is a living document that will be updated as new information is learned that can help guide the risk assessment process.

Project timing: The USACE and/or BOEM will determine project timing and necessary minimization measures to reduce the risk of take of ESA-listed species through the Risk Based Adaptive Management process outlined in Section 2.9.2.2 of the 2020 SARBO and Appendix J. Additional timing requirements apply within the range of certain species, as outlined in the North Atlantic Right Whale Conservation Plan (Appendix F) and sturgeon PDCs (Appendix E).

Additional regulations that may apply: Note that additional state and federal requirements may apply to projects covered under the 2020 SARBO. This may include, but is not limited to, requirements under the Magnuson–Stevens Fishery Conservation and Management Act, Marine Mammal Protection Act, Clean Water Act, National Marine Sanctuaries Act, and/or state-specific regulations.

Appendix B. 2020 SARBO General PDCs

1 Eligibility Criteria PDCs: PDCs that Define if an Activity Meets the Criteria to be Covered under the 2020 SARBO

The Eligibility PDCs in this section define the criteria that must be met for different types of activities to be covered under the 2020 SARBO. All activities must follow the 2020 SARBO PDCs to be covered under this Opinion. Section 2 of the 2020 SARBO provides additional information, definitions, and images to help explain each of these types of projects.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Substantially Similar Projects or Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but are substantially similar, or projects with substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the Opinion, will require individual Section 7 consultation. In addition, any area previously authorized or permitted to be dredged or have material placed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

1.1 Dredging and dredging related activities covered under the 2020 SARBO

DREDGE.1 Maintenance dredging covered under this Opinion includes the list below, as described in 2.3.1 of the 2020 SARBO.

- Maintenance dredging in navigation waterways and channels required to be maintained under Title 33 (Navigation and navigable waters): Maintenance to the dredge template provided in Title 33 or the deeper or wider template provided in the SARBO Biological Assessment (SARBA) Appendix B (provided on the NMFS dredging website at https://www.fisheries.noaa.gov/content/southeast-dredging) or analyzed in an individual Section 7 consultation, including the defined overdepth and advanced maintenance depth.
- Maintenance dredging in navigation channels (not required to be maintained under Title 33): Maintenance to the dredge template provided in in SARBA Appendix B or to the dredge template federally authorized or permitted and previously dredged. The dredging template includes the overdepth and advanced maintenance depth analyzed in a consultation during the evaluation of the previous dredging event.
    - o Maintenance activities should occur at a frequency such that the area is navigable, barring a sudden change from a storm, and that returning the area to the authorized or permitted dredge template does not alter the hydrology of the area. For example, dredging a channel that has not been

maintained and gradually returned to the surrounding conditions, is not considered maintenance.

  - Maintenance dredging in navigation channels other than the main federal channels, such as the secondary channel sections of a braided river that is not part of the main channel, or a channel/canal that connects the main navigation channel to coastal communities and/or coastal neighborhoods.
    - If another programmatic exists that covers this action, the other regional programmatic will be used, such as the programmatic the *Biological Opinion on the authorization of minor in-water activities throughout the geographic area of jurisdiction of the U.S. Army Corps of Engineers Jacksonville District, including Florida and the U.S. Caribbean (JAXBO)* (NMFS tracking number SER-2015-17616) that covers minor and maintenance dredging in Florida.[85]

- <u>Maintenance dredging areas other than navigation channels</u>: Maintenance dredging of an area to the previously authorized dredge template, as further specified below. Maintenance dredging in areas other than navigation channels may include:
  - Maintenance dredging ports and berths along maintained navigation channels including those not owned and operated by a Port Authority.
  - Maintenance dredging in smaller areas such as public and private marinas, boat ramps, and around docks.
- <u>Maintenance of sediment traps</u>: Maintenance of existing sediment traps to the previous dredge template.
- <u>Minor channel modifications, realignment, or bend easing</u>: Minor channel modifications considered under this Opinion are limited to minor realignments that follow the naturally shifting deep water channel to the same depth and width as the previously maintained channel or realignment of an existing channel that shifted. Intentional minor realignment (e.g. bend easing) is not covered.

DREDGE.2 Borrow area dredging covered under this Opinion includes existing sites and new borrow sites that meet the conditions listed below and described in Section 2.3.2 of the 2020 SARBO. Borrow area dredging is limited to a depth that does not result in hypoxic or anoxic conditions in the area. Hypoxic conditions are those with reduced dissolved oxygen (DO) and anoxic refers to areas with little to no remaining DO needed for most aquatic life to survive. Examples of dredging that may result in these conditions include the digging of step banked, deep holes that prevent water exchange. This will be determined by the USACE and/or BOEM based on modeling and past experience dredging in the area.

[85] https://www.fisheries.noaa.gov/webdam/download/91825944

DREDGE.3 Agitation dredging covered under this Opinion includes bed-leveling and water-injection dredging used as a form of maintenance dredging as described in 2.5.3 of the 2020 SARBO.

DREDGE.4 Environmental Restoration (e.g., Muck) Dredging. Muck dredging is covered under this Opinion to the dredging depths necessary to remove the organic/muck layer down to natural sediments and does not remove the natural (non-muck) sediments below, to the maximum extent practical based on dredging precision. It cannot be used to increase water depths to support navigation, access, or vessel mooring.

1.2 Dredged material placement covered under the 2020 SARBO

The following categories of in-water placement are included and described in Section 2.4 of the 2020 SARBO.

PLACE.1 Beneficial use (e.g., beach nourishment, nearshore placement, or muck dredging considered under 2020 SARBO or marsh creation locations analyzed under a separate ESA Section 7 consultation, but filled with material dredged under 2020 SARBO)

- Beach nourishment described in Section 2.4.1 2020 SARBO and PDC PLACE.2.
- Nearshore placement described in Section 2.4.2 2020 SARBO and PDC PLACE.3.
- Beneficial use placement of material where the dredging of the material is covered under this Opinion and placement of material in a specific location was analyzed under an individual Section 7 consultation (e.g., placement of material used in marsh creation).
- Beneficial use activities not covered include thin-layer placement (e.g., used for marsh creation or other disposal method), filling of holes to improve water quality, filling of holes or minor depressions to restore the appropriate depth for habitat restoration, or other similar placement activities.

PLACE.2 Beach nourishment projects are covered under this Opinion if they meet the conditions listed below and described in Section 2.4.1 of the 2020 SARBO.

- Beach nourishment in the locations and defined beach sand placement template described in SARBA Appendix B.
- Beach nourishment in areas that has been previously analyzed in a Separate Section 7 consultation, filled, and is being nourished again to the same beach sand placement template.
- Placement on the uplands for activities with no intended equilibrium to occur in water (e.g., dune restoration) is outside of the jurisdiction of NMFS.
- No beach nourishment projects are covered in the U.S. Caribbean.
- New beach nourishment and placement is allowed outside the range of corals (as defined in the Coral PDCs in Appendix C) if it meets the conditions

below. For the purposes of this Opinion, new beach placement is defined as placement of sand on an existing beach that has not been previously nourished.

- o Placement of beach sand outside of Florida will be compatible with the native beach sediment composition to minimize turbidity in the surrounding in-water environment.
- o New beach placement is allowed if the design profile is similar/consistent to adjacent beaches. This does not include non-traditional beach nourishment designs such as those that protrude and may obstruct species movement along the shore.
- o All new beach nourishment is limited to placement in areas lacking hardbottom (e.g., worm-rock or other forms of non-coral hardbottom) and seagrasses that may be used as foraging or refuge habitat for ESA-listed species.

PLACE.3 Nearshore placement is covered under this Opinion that meet the conditions listed below and described in Section 2.4.2 of the 2020 SARBO.

- Nearshore placement described in SARBA Appendix B, which is generally related to beach nourishment projects.
- Nearshore placement in areas that have undergone an individual Section 7 consultation and require repeat placement within the same area.
- New nearshore placement adjacent to beaches, through the use of side-casting material adjacent to a dredge location, or any other placement in water is allowed outside the range of Johnson's seagrass (Johnson's Seagrass PCDs, Appendix D), outside the range of ESA-listed corals (Coral PDCs, Appendix C), and outside of sturgeon rivers (Sturgeon PDCs, Appendix E).

PLACE.4 Ocean dredged material disposal site (ODMDS) is limited to the locations provided in SARBA Appendix B or ODMDS locations that have been analyzed under an individual ESA Section 7 consultation prior to construction. New ODMDS locations are not otherwise covered.

PLACE.5 Temporary offshore staging area for the purpose of future use of the placement material, typically occurring within an existing ODMDS or within a location previously approved for offshore placement of dredged material.

PLACE.6 Upland Placement, which is defined as placement not occurring in a natural body of water and outside of NMFS purview, must meet the following criteria:

- Upland placement projects with return/discharge water to waters under NMFS purview will be designed to assure that turbidity generated by the discharge waters has returned to ambient levels before reaching any nearby ESA-listed coral or Johnson's seagrass.
- Discharge flow will be maintained to prevent scour or erosion.

1.3 Transportation of dredge material

Materials that are dredged or placed during activities that are covered under this Opinion can be transported using the equipment described in Section 2 of the 2020 SARBO. Equipment specific and location specific PDCs are provided in the 2020 SARBO PDCs.

1.4 Geophysical and Geotechnical (G&G) Surveys

G&G surveys, as described in Section 2.6 of 2020 SARBO, may be used to determine sediment composition and depth in areas where dredging or material placement can occur under 2020 SARBO. G&G surveys may also be used to identify sensitive resources in areas surrounding the areas proposed for dredging, or material placement such as hardbottom habitat within the range of ESA-listed corals (Coral PDCs, Appendix C), or areas of seagrass within the range of Johnson's seagrass (Johnson's seagrass PDCs, Appendix D).

2 Standard/General PDCs: PDC Requirements that Apply to All Projects Covered under the 2020 SARBO

This section provides general PDCs that apply to all projects covered under the 2020 SARBO.

2.1 Education and observation requirements

These PDCs are designed to educate all on-site personnel including the vessel captain, crew, and PSO of the requirements to avoid and minimize effects to ESA-listed species and critical habitat. All personnel must be made aware of and adhere to them.

EDUCATE.1 The USACE and BOEM must ensure that all personnel associated with projects authorized under 2020 SARBO are instructed about the potential presence of species protected under the ESA and MMPA and the appropriate protocols if they are encountered including those in the PSO PDCs in Appendix H.

EDUCATE.2 All on-site project personnel are responsible for observing water-related activities for the presence of ESA-listed species.

EDUCATE.3 All on-site project personnel will be informed of all ESA-listed species that may be present in the area and advised that there are civil and criminal penalties for harming, harassing, or killing ESA-listed species or marine mammals.

EDUCATE.4 All on-site project personnel will be briefed that the disposal of waste materials into the marine environment is prohibited. All crew will attempt to remove and properly dispose of all marine debris discovered during dredging operations, to the maximum extent possible.

2.2 In-water dredging and material placement requirements

The PDCs in this section apply to all in-water activates, if applicable.

INWATER.1 Species Movement: All work, including equipment, staging areas, and placement of materials, will be done in a manner that does not block access of ESA-listed species from moving around or past construction.

- Sand placed on the beach or in the nearshore littoral areas will be placed in a manner that does not create mounds or berms that could prevent nesting sea turtles or hatchings from entering or exiting the beach from nearshore waters.
- All placement, including ODMDS placement, will not create an obstruction of species movement in the area (e.g., does not create a mound that would deter or prevent species from moving through the area).

INWATER.2 Equipment placement: Equipment will be staged, placed, and moved in areas and ways that minimize effects to species and resources in the area, to the maximum extent possible. Specifically:

- All vessels will preferentially follow deep-water routes (e.g., marked channels) to avoid potential groundings or damaging bottom resources whenever possible and practicable.
- If barges, scows, and other similar support equipment are used, they will be positioned away from areas with sensitive bottom resources such as non-ESA-listed seagrasses, corals, and hardbottom, to the maximum extent possible.
- If pipelines are used, they will be placed in areas away from bottom resources and of sufficient size or weight to prevent movement or anchored to prevent movement or the pipeline will be floated over sensitive areas.

INWATER.3 Turbidity control: All work that may generate turbidity will be completed in a way that minimizes the risk of turbidity and sedimentation reaching non-mobile ESA-listed species (i.e., ESA listed corals and Johnson's seagrasses) as well as other non-ESA-listed non-mobile species (e.g., non-ESA-listed corals, sponges, and other natural resources) to the maximum extent practicable. This may include selecting equipment types that minimize turbidity and positioning equipment away or downstream of non-mobile species.

INWATER.4 Turbidity curtains: Turbidity curtains may be used to maintain water quality standards where appropriate and practicable with consideration given to ambient turbidity and if the curtains are practical based on current, wave action, or other factors.

- If turbidity curtains are used, barriers will be positioned in a way that does not block species' entry to or exit from designated critical habitat and does not entrap species within the construction area or block access for them to navigate around the construction area.
- Project personnel must take measures to monitor for entrapped species in areas contained by turbidity curtains and allow access for them to escape if spotted.

- Beach nourishment projects will be designed to minimize turbidity in nearshore waters by using methods that promote settlement before water returns to the water body (i.e., shore parallel dikes). Turbidity and marine sedimentation will be further controlled using land-based erosion and sediment control measures to the maximum extent practicable. Land-based erosion and sediment control measures will (1) be inspected regularly to remove excess material that could be an entanglement risk, (2) be removed promptly upon project completion, (3) and will not block entry to or exit from designated critical habitat for ESA-listed species.

INWATER.5 Entanglement: If lines or cables are used (e.g., to mark floating buoys, lines connecting pickup buoy lines, or for turbidity curtains):

- In-water lines (rope, chain, and cable) will be stiff, taut, non-looping. Examples of such lines are heavy metal chains or heavy cables that do not readily loop and tangle. Flexible in-water lines, such as nylon rope or any lines that could loop or tangle, will be enclosed in a plastic or rubber sleeve/tube to add rigidity and to prevent the line from looping or tangling. In all instances, no excess line is allowed in the water. Requirements for lines associated with relocation trawling are handled separately in Appendix B Section 3.5.
- All lines or cables will be immediately removed upon project completion.
- All in-water line and materials will be monitored regularly to ensure nothing has become entangled.
- Cables or lines with loops used to move pipelines or buoys will not be left in the water unattended.

INWATER.7 Dredging or material placement in areas not previously used for dredging or placement are allowed under this Opinion for borrow sites, side-cast dredging, beach nourishment, nearshore placement associated with beach nourishment, if they meet all of the PDCs in this Opinion, including those listed below:

- Within the range of ESA-listed corals (Coral PDCs in Appendix C), within the range of Johnson’s seagrass (Johnson’s Seagrass PDCs in Appendix D), and in sturgeon rivers (Sturgeon PDCs in Appendix E): Additional PDCs apply to these activities.
- Within the range of Cape Canaveral to Miami-Dade, Florida and in the U.S. Caribbean: Areas not previously dredged or areas where in-water material placement has not previously occurred will not remove or place materials on nearshore or surf-zone, low-profile hardbottom outcroppings (e.g., worm-rock reef [sabellariid worm reefs] and eolianite, granodiorite). This habitat can be persistent or ephemeral, cycling through periods of exposure and cover by sand. Continued maintenance of existing beach nourishment projects must have already considered protection of this habitat when calculating the equilibrium toe of fill (ETOF).

- Within 400 ft of any significant non-coral hardbottom areas or bottom structures that serve as attractants to sea turtles for foraging or shelter: For purposes of the 2020 SARBO, NMFS considers significant non-coral hardbottom to be an area with a horizontal distance of 150 ft that has an average elevation above the sand of 1.5 ft or greater and has algae growing on it. BOEM and USACE Districts will identify any non-coral hardbottom prior to commencing work to ensure that all sand removal projects provide a 400 ft buffer from all equipment. If BOEM and/or the USACE is uncertain as to what constitutes significance, it/they will coordinate with NMFS Southeast Region Habitat Conservation Division (727-824-5317) and NMFS' Protected Resources Division (727-824-5312) for clarification and guidance. Walls of federally-maintained navigation channels (i.e., jetties and other such man-made structures) are not considered hardbottom for the purpose of this 2020 Opinion.
- In areas with seagrass: Dredging and placement in new areas will avoid areas with non-listed seagrasses to the maximum extent practicable.

INWATER.8 Lighting near sea turtle nesting beaches: For dredges and any support vessels operating at night in front of nesting beaches, lighting will be limited to the minimal lighting necessary to comply with U.S. Coast Guard and Occupational Safety and Health Administration requirements (most up-to-date version of Engineering Manual 385-1-1). Lighting associated with beach nourishment construction activities will be minimized through reduction, shielding, lowering, and/or use of turtle friendly lights, to the extent practicable without compromising safety, to reduce potential disorientation effects on female sea turtles approaching the nesting beaches and sea turtle hatchlings making their way seaward from their natal beaches. As technology changes, so do turtle friendly lighting options. New information/technology should be used as soon as published guidance for types of appropriate lights and appropriate shielding and positioning of lights is available that is protective of sea turtles (e.g., those outlined by the Florida Fish and Wildlife Conservation Commission's website http://myfwc.com/wildlifehabitats/managed/sea-turtles/lighting/).

3 Equipment-Specific PDCs for Projects Covered under the 2020 SARBO

This section provides equipment-specific PDCs including requirements that must be met when designing a project with work occurring using the equipment types discussed in Section 2.5 of the 2020 SARBO. All on-site project personnel, including the vessel captain, crew, and PSO, must be aware of and adhere to these requirements.

3.1 Hopper dredge requirements

HOPPER.1 During all hopper dredging operations, NMFS-approved PSOs will monitor for the presence of ESA-listed species. The dredge operator will maintain a safe working environment for the PSO to access and effectively monitor inflow screening, overflow screening, and dragheads for incidental take of ESA-listed species and associated bycatch after every load. All new hopper dredge vessels or modifications made to existing vessels must be designed to allow safe access to and/or visibility of all collected material in both the inflow box and overflow screening areas so that the PSO is able to inspect the contents after every load for evidence of ESA-listed species. The appointed contact (e.g., Quality Assurance Representative or the Contractor) will immediately notify the USACE who will notify the SARBO Team if conditions limit the ability to safely monitor dredging operations.

Draghead Observation:
Upon completion of each load cycle, dragheads will be monitored as the draghead is lifted from the sea floor and placed on the saddle in order to assure that ESA-listed species that may be impinged within the draghead are observed and accounted for. The PSO, or designated dredge crew member under the guidance and supervision of the PSO when safety is of concern, must physically inspect dragheads for evidence of ESA-listed species take after every load.

Inflow screening Observation:

- Inflow screening must be designed to capture and retain material for the PSO to monitor for the presence of ESA-listed species. The screened area must be accessible to the PSO to ensure 100% observer coverage. The PSO must inspect the contents of all inflow screening boxes after every load, including opening the box (where applicable and safely accessible) and looking inside at all contents for evidence of ESA-listed species entrainment. If the contents are not clearly visible and identifiable from a location outside of the box, then in limited instances, the PSO may be required to enter the inflow box to identify contents for evidence of ESA-listed species take.
- All hopper dredges are required to have 100% inflow screening unless they must be removed for safety due to clogging as outlined below.
    - Inflow screening size will start at 4-inch by 4-inch, but may be gradually adjusted to a larger screen size if clogging reduces the ability for the PSO to monitor the inflow for the presence of ESA-listed species or if clogging reduces dredging production and thereby expands the time dredging is

required. Scenarios that may result in the clogging of inflow and overflow screens are dredge and project specific.

  - All modifications will be made in close coordination with the dredging contractor, PSO, appropriate USACE and/or BOEM project managers, and NMFS. The USACE and/or BOEM will provide NMFS with a notification when screen sizes are increased or inflow screens are removed that will include an explanation of what attempts were made to reduce the clogging problem, how long the problem may persist, and how effective overflow screening will be achieved.
  - If inflow screens are increased to be larger than 4-inch by 4-inch or are removed due to clogging, the USACE and/or BOEM will continue to re-evaluate the risk of clogging on a load by load basis and the inflow screens will be reinstated when clogging is no longer occurring. The USACE will track the number of loads that inflow screens were removed as part of the reporting requirements in Section 2.9.3.5 of the 2020 SARBO.

- Hopper dredge operators will not open the hydraulic doors on the inflow boxes prior to inspection by the PSO for evidence of ESA-listed take.
- If the inflow box cannot be observed due to clogging, the box contents cannot be dumped or flushed unless overflow screening that captures contents for observation by the PSO is operational and monitored for evidence of take. Once overflow screening is operational, PSOs shall also visually monitor box contents as they are dumped or flushed into the hopper.

Overflow Screening Observations

- All hopper dredges are recommended to have operational overflow screening and monitor for take after each load. Overflow screening is required to be installed and monitored after each load if the inflow screening is removed or bypassed due to clogging.
- Overflow screening must be designed to capture and retain material larger than the screen size for the PSO to monitor for the presence of ESA-listed species. The screened area must be accessible to the PSO to inspect for evidence of ESA-listed species take.
- Screen size will start at 4-inch by 4-inch, but may be adjusted to a larger screen size if clogging reduces the ability for the PSO to monitor the screen for the presence of ESA-listed species or if clogging reduces dredging production and thereby expands the time dredging is required. All modifications will be made in close coordination with the dredging contractor, PSO, appropriate USACE and/or BOEM project managers, and NMFS. If screen sizes are increased due to clogging, the risk of clogging will be re-evaluated weekly and the overflow screens will be reinstated using the smallest screen size that can be effectively used (preferably 4 inch by 4 inch) when clogging is no longer occurring.

HOPPER.2 To prevent impingement or entrainment of ESA-listed species within the water column, dredging pumps will be disengaged by the operator when the dragheads are not actively dredging and therefore working to keep the draghead firmly on the bottom. Pumps will be disengaged when lowering dragheads to the bottom to start dredging, turning, or lifting dragheads off the bottom at the completion of dredging. Hopper dredges may utilize a bypass or other system that would allow pumps to remain engaged, but result in no suction passing through the draghead. This dredge modification (when employed) is commonly referred to as a turtle bypass valve. This precaution is especially important during the cleanup phase of navigation dredging operations to remove remaining high spots or when a shallow veneer of compatible sediment remains within a borrow area; thus limiting overdepth dredging and plowing efficacy of the turtle deflector. In these example circumstances, the draghead may frequently come off the bottom and can suck in turtles/sturgeon resting or foraging in shallow depressions.

HOPPER.3 Pumping water through the dragheads is not allowed while maneuvering or during travel to/from the disposal or pumpout area. The dredge operator will ensure the draghead is embedded in sediment when pumps are operational, to the maximum extent practicable.

HOPPER.4 All waterport or other openings on the hopper dredge are required to be screened to prevent ESA-listed species from entering the dredge.

HOPPER.5 A state-of-the-art solid-faced deflector that is attached to the draghead must be used on all hopper dredges at all times.

3.2 Munitions and explosives (MEC)/ unexploded ordinance (USO) screening on a draghead requirements

MEC screening is used in areas where munitions may be present, but may also be used for other purposes such as handling areas with rock. If MEC screening is used, the following PDCs apply:

MEC.1 Prior to any use of MEC/ UXO screens on a project, the use of this screening will be reviewed by NMFS through the alternative review/ supersede process outlined in Section 2.9.5 of the 2020 SARBO.

MEC.2 The PSO will be required to inspect the draghead MEC screens after every load to verify that no ESA-listed species are impinged on the screening.

MEC.3 If MEC screening is used on a beach nourishment outflow, screening will be monitored and USACE and/or BOEM will be notified of any potential ESA-listed species takes identified in the beach outflow screening box.

3.3 Cutterhead Dredge

CUTTER.1 The cutterhead will not be engaged/turned on when not embedded in the sediment, to the maximum extent possible.

3.4 Bed-leveling requirements

LEVEL.1 Bed-levelers used as part of the proposed action will be of a design that produces a sand wave in front of the leading face of the bed-leveling device such that it disturbs sea turtles off the sea/channel floor bottom. All support structures must be welded to prevent impingement or "pinch points" for passing ESA-listed species. The design analyzed in the Brunswick Harbor study is approved to meet these requirements (Dodd 2003). Any other design must be documented and photographed and submitted with the pre-construction notification and during the annual review outlined in Section 2.9 of the 2020 SARBO in order to monitor the designs used. Additional designs may be deemed acceptable during the annual review.

LEVEL.2 The bed-leveler will be slowly lowered to the sea/channel bottom and the depth of the bed-leveler adjusted constantly to meet required depth and to compensate for tidal fluctuations.

LEVEL.3 The bed-leveler will be towed/pushed along the bottom no faster than needed to move the material at the sea/channel bottom (approximately 1-2 knots).

3.5 Relocation trawling requirements

The relocation trawling requirement describes the type of trawling allowed. Handling and reporting of ESA-listed species captured are provided in the PSO PDCs in Appendix H.

RELOCATE.1 USACE and/or BOEM are authorized and will utilize relocation trawling and/or non-capture trawling in association with dredging activities in reasonable circumstances as an avoidance and minimization measure to reduce the risk of potential lethal take of ESA-listed species.

RELOCATE.2 If relocation trawling is deemed appropriate to minimize the risk of lethal take on a project using the risk assessment process outlined in Section 2.9.2.2 of the 2020 SARBO, trawlers will mobilize as quickly as possible.[86]

RELOCATE.3 Trawling specifications listed below and in the PSO PDCs in Appendix H will be followed.

- Trawl tow-time duration will not exceed 42 minutes (doors in - doors out).
- Trawl speeds will not exceed 3.5 knots for normal operations; however, speeds may be increased to the minimum speed needed to maintain control of the vessel.

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[86] USACE and BOEM report that response time for relocation trawlers is typically 48-72 hours. Flexibility to ensure the most appropriate relocation trawler for the species/location is chosen may increase the response time.

- Lazy lines will be designed according to the design specifications in Appendix I to minimize the risk of entanglement with captured species.

RELOCATE.4 Trawling within the range of ESA-listed corals (defined in Appendix C as Palm Beach County, Florida south through the Florida Keys and in the U.S. Virgin Islands and Puerto Rico) is not covered under this Opinion.

RELOCATE.5 Relocation trawling is not covered under this Opinion in the U.S. Caribbean (i.e., U.S. Virgin Islands and Puerto Rico)

Appendix C. 2020 SARBO Coral PDCs

The PDCs in this appendix apply to all projects that occur within the range of ESA-listed corals, as defined in in this appendix. These requirements are in addition to any other applicable PDCs outlined in in the 2020 SARBO.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Substantially Similar Projects or Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but are substantially similar, or projects with substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the Opinion, will require individual Section 7 consultation. In addition, any area previously authorized or permitted to be dredged or have material placed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

1 Description of the Areas Coral PDCs Apply

Coral PDC Section 1.1 provides information on *Acropora* critical habitat, designated to protect critical habitat for elkhorn and staghorn corals. The 5 other ESA-listed coral species (boulder star, lobed star, mountainous star, pillar, and rough cactus coral) may occur in the 2020 SARBO action area, but NMFS has not designated critical habitat for those species. Coral PDC Section 1.2 defines the geographic range of all ESA-listed corals in which adherence to the Coral PDCs is required by the 2020 SARBO.

1.1 *Acropora* Critical Habitat

According to the Final Rule designating *Acropora* critical habitat (73 FR 72210, Publication Date November 26, 2008), the physical feature essential to the conservation of elkhorn and staghorn corals is: substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. "Substrate of suitable quality and availability" is defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover.

The Final Rule designated 4 specific areas of critical habitat:

1. the Florida area, which comprises approximately 1,329 square miles (3,442 km^2) of marine habitat;
2. the Puerto Rico area, which comprises approximately 1,383 square miles (3,582 km^2) of marine habitat;
3. the St. John/St. Thomas area, which comprises approximately 121 square miles (313 km^2) of marine habitat;

4. the St. Croix area, which comprises approximately 126 square miles (326 km²) of marine habitat.

Figure 49 and Figure 50 provide images of critical habitat, and geographic information system (GIS) data layers of critical habitat maps are available for download on the NMFS website at https://sero.nmfs.noaa.gov/maps_gis_data/index.html.

Note the shoreward boundary is the 6-ft (1.8 m) contour from Boynton Inlet to Government Cut, Miami-Dade County and is mean low water line in all other areas. Assessment of project effects on critical habitat does not consider the omitted areas presented in the Final Rule designating critical habitat (73 FR 72209, Publication Date November 26, 2008), and described in Coral PDC Section 1.1.1 below.

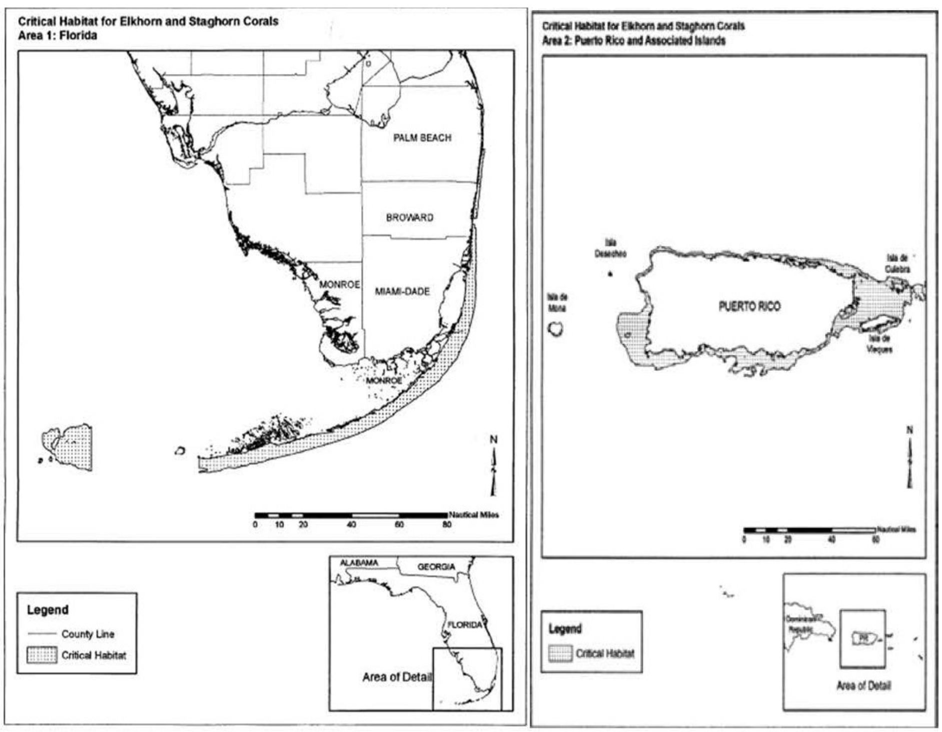


Figure 49. The left image is for *Acropora* critical habitat Area 1 (Florida Unit) and the right image is for Area 2 (Puerto Rico and Associated Islands).

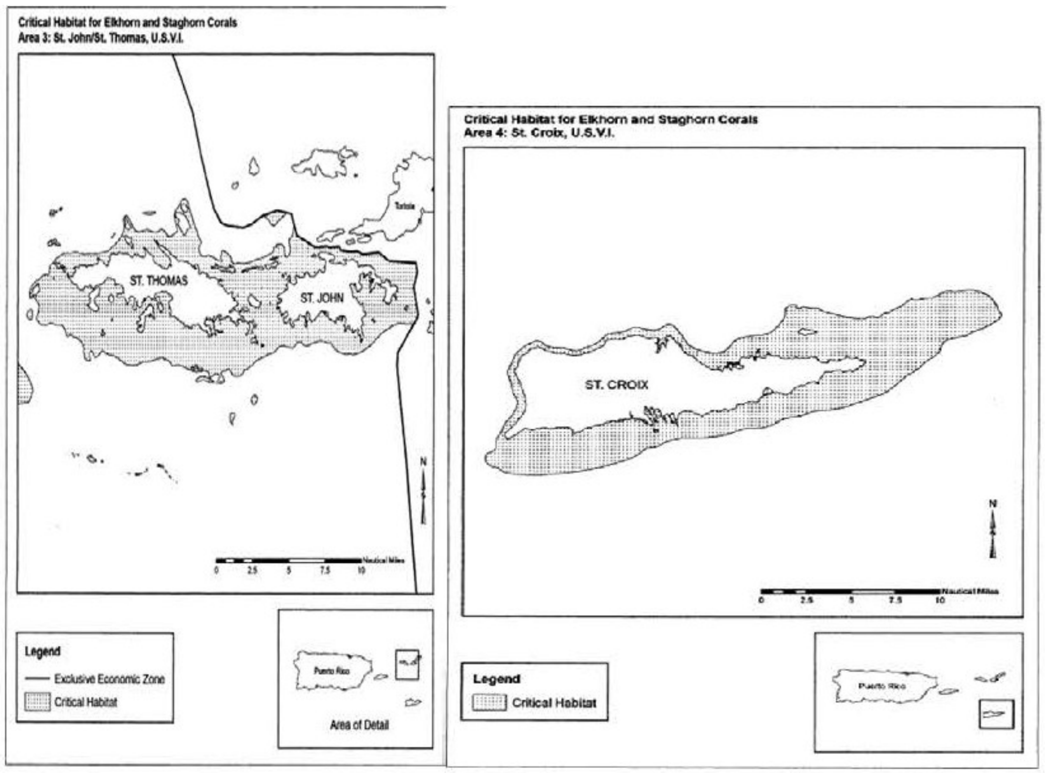


Figure 50. The left image is for *Acropora* critical habitat Area 3 (St. Thomas/St John, U.S. Virgin Islands Unit) and the right image is for Area 4 (St. Croix, U.S. Virgin Islands Unit).

1.1.1 Areas Omitted from *Acropora* Critical Habitat

As defined in the Final Rule (73 FR 72209, Publication Date November 26, 2008), *Acropora* critical habitat does not include the following particular areas where they overlap with the areas described above:

1. All areas subject to the 2008 Naval Air Station Key West Integrated Natural Resources Management Plan.
2. All areas containing existing (already constructed) federally authorized or permitted man-made structures such as aids-to-navigation, artificial reefs, boat ramps, docks, pilings, maintained channels, or marinas.
3. All waters identified as existing (already constructed) federally authorized channels and harbors as follows:
   (i) Palm Beach Harbor; (ii) Hillsboro Inlet; (iii) Port Everglades; (iv) Miami Harbor; (v) Key West Harbor; (vi) Arecibo Harbor; (vii) San Juan Harbor; (viii) Fajardo Harbor; (ix) Ponce Harbor; (x) Mayaguez Harbor; (xi) St. Thomas Harbor; and (xii) Christiansted Harbor.

In addition to the above, 1 military site known as the Dania Restricted Anchorage Area, comprising approximately 5.5 mi² (14.3 km²), excluded from critical habitat because of national security impacts. This excluded area is represented by the break in *Acropora* critical habitat that is shaded in pink in Figure 51 below.

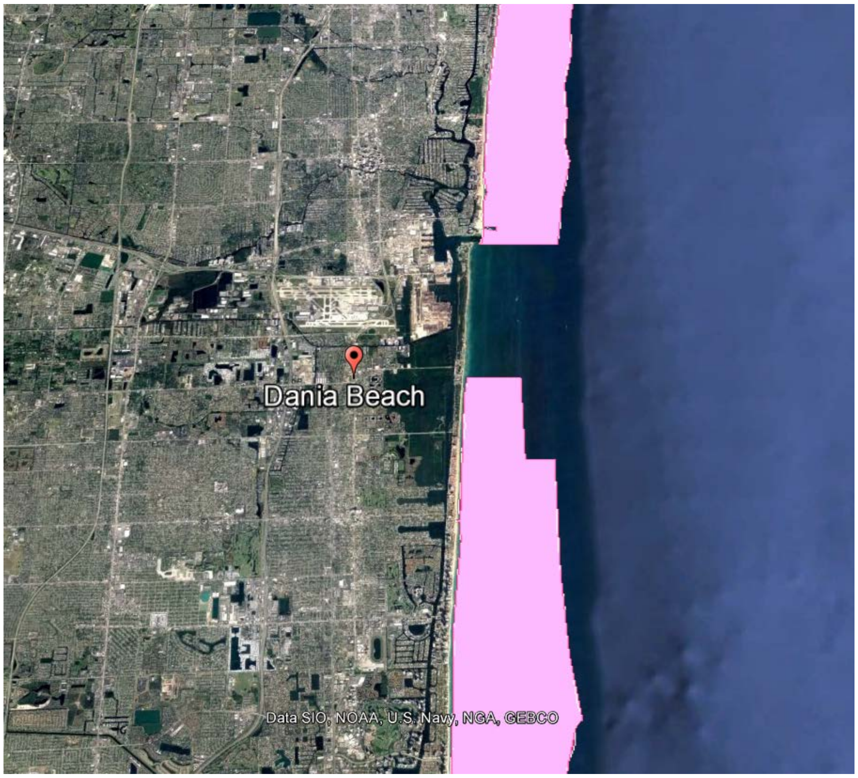


Figure 51. *Acropora* critical habitat exclusion in the Dania restricted anchorage area shown as the break in the *Acropora* critical habitat area shaded pink.
Image from © 2018 Google, data SIO, NOAA, U.S. Navy, NGA, GEBCO.

1.2 Areas and terms defined for the 2020 SARBO

1.2.1 Definition of the range of ESA-listed corals

For the purposes of the 2020 SARBO, the range of ESA-listed corals is defined as all areas from the St. Lucie Inlet in Martin County, Florida south through the Florida Keys, Puerto Rico and the U.S. Virgin Islands from mean low water line to 262 ft (80 m) depth. While the range of ESA-listed corals includes the area designated as *Acropora* critical habitat, the Coral PDCs encompass a larger area in order to be protective of the entire range where ESA-listed corals may be present.

1.2.2 Coral Hardbottom

Corals may grow on any hard surface including both natural, consolidated hard substrate and man-made structures, such as seawalls, groins, jetties, bulkheads, dock pilings, and aids to navigation, within the range of corals. For the purposes of the 2020 SARBO and consistent with the Final Rule designating *Acropora* critical habitat (73 FR 72210, Publication Date November 26, 2008), only natural substrate is considered to contain the essential habitat feature necessary to support ESA-listed corals. Areas containing this habitat feature within the range of ESA-listed corals will be referred to as coral hardbottom throughout the rest of this document.

For purposes of the 2020 SARBO, coral hardbottom is defined in the same way as the essential feature for *Acropora* critical habitat: as substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. "Substrate of suitable quality and availability" is defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover.

For purposes of the Coral PDCs, we use the presence of coral hardbottom as a way to identify areas where ESA-listed corals may be found. This includes:

- All areas within the range of ESA-listed corals (defined in Coral PDC Section 1.2.1 which includes, but is not limited to, *Acropora* critical habitat) that has substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments (as defined in the *Acropora* critical habitat rule).
- Areas excluded by the *Acropora* critical habitat Final Rule (Coral PDC Section 1.1.1) because they lack the essential habitat feature are also excluded as coral hardbottom habitat for purposes of this Opinion. It is important to note that ESA-listed coral colonies may be located in areas excluded from *Acropora* critical habitat (e.g., on man-made structures or natural consolidated substrate in excluded areas), and effects to corals in these areas will be considered in this Opinion. The Coral PDCs do not require surveying for or reporting of corals growing on surfaces other than coral hardbottom (e.g., on man-made structures).
- Areas containing the essential habitat feature that were excluded from *Acropora* critical habitat are not excluded as coral hardbottom habitat for the purposes of this Opinion. The Coral PDCs require surveying for or reporting of corals growing on coral hardbottom containing substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments.
- For the purposes of this Opinion, coral hardbottom extends within the range of corals in Florida from the 6-ft (1.8 m) contour (waterward of the beach, shore, or inlet) to the 262-ft (80 m) contour and in the U.S. Caribbean from the mean low water line to 262-ft (80 m) contour. While *Acropora* critical habitat identified the depth range for *Acropora* corals to extend to only 30 m, other ESA-listed corals can be found up to 80 m deep.

1.3 Hardbottom survey area

The Coral PDCs require surveying to identify the presence of coral hardbottom as a protection for both *Acropora* critical habitat feature and as a simplified way to identify areas where ESA-listed corals may occur. The areas, distances, and survey methods required to identify coral hardbottom located near the dredge or beach nourishment projects covered under the 2020 SARBO are provided in the Coral PDCs.

Once coral hardbottom is identified based on the hardbottom surveys, additional measures may be required to complete the project including limiting certain types of equipment used, restricting the length of time construction can occur near coral hardbottom, or relocating the ESA-listed corals in the hardbottom area, as described by project type in the Coral PDCs.

2 Requirements for All Dredge and Material Placement Projects Within the Range of ESA-listed Corals

The following PDCs apply to all projects within the defined range of ESA-listed corals (Coral PDC Section 1.2.1 above). These PDCs are in addition to any other applicable PDCs provided in the 2020 SARBO.

2.1 Equipment and Surveying

The following PDCs apply to all projects within the defined range of ESA-listed corals that include channel and borrow area dredging.

CORAL.1 All vessel anchoring and spudding is limited to unconsolidated and uncolonized areas (i.e., sand areas lacking coral hardbottom and uncolonized by corals).

CORAL.2 Dredging sediment composition
Sediment type in dredge areas will be surveyed prior to dredging by employing a scientific sampling survey that provides a representative sample of the sediment from all areas of the dredge project footprint. Samples will be collected within 2 years prior to dredging of navigation channels and 5 years prior to dredging in borrow areas. Samples will be sent for a laboratory analysis of sediment grain size.

2.2 Dredging within the range of ESA-listed Corals

CORAL.3 Dredging that requires the penetration of rock or other hard substrate is not allowed.

CORAL.4 Emergency dredging of navigation channels
Maintenance channel dredging within the range of ESA-listed corals that is required after a natural disaster will be handled under the emergency consultation process[87] if the work performed is completed within 2 months of the natural disaster. Emergency consultation procedures are outlined in the NMFS website at https://www.fisheries.noaa.gov/content/emergency-consultations-southeast. If the maintenance dredging begins more than 2 months after the natural disaster, the dredging will follow the requirements of the 2020 SARBO to minimize the additional effects to ESA-listed corals and *Acropora* critical habitat analyzed in the 2020 SARBO.

[87] The regulations regarding ESA Section 7 consultations for emergency circumstances such as situations involving acts of God, disasters, casualties, national defense or security emergencies, etc., allow for response activities that must be taken to prevent imminent loss of human life or property (50 CFR 402.05 (a)).

CORAL.5 Maintenance dredging of navigation channels and dredging in borrow areas.

- The type of dredging allowed based on the equipment type, sediment type that will be dredged (PDC CORAL.2), dredging time limits, and proximity of work to coral hardbottom (defined in Coral PDC Section 1.2.2) is listed in Table 54.
- Any dredging within the range of corals that will dredge material over 10% fines is not covered under this Opinion, except within the semi-enclosed portions of the Port of Miami Harbor, Port Everglades, and San Juan Harbor defined below.
- The semi-enclosed portions of the Port of Miami Harbor, Port Everglades, and San Juan Harbor defined below are NOT subject to the limitations in Table 54 within the ports/harbor areas shoreward of the line formed by the Global Positioning System (GPS) points provided below. Dredging in these areas may be done by any equipment type and of material with any percent fines, in compliance with all other relevant PDCs. The USACE will minimize turbidity to the maximum extent practicable to ensure turbidity does not result in sedimentation cover of corals outside of the port or harbor.
  - Port of Miami 25.7642444°N, 80.1307306°W and 25.7623889°N, 80.1337694°W
  - Port Everglades 26.0955167°N, 80.1056694°W and 26.0925139°N, -80.1081694°W
  - San Juan Harbor 18.4508306°, 66.1289278° and 18.4588917°N, 66.1166083°W

CORAL.6 High-resolution geophysical surveys sufficient to detect and delineate any hardbottom areas will be used to fulfill hardbottom identification requirements in the Coral PDCs. These surveys will be conducted within 2 years prior to channel dredging or beach nourishment projects and within 5 years prior to borrow site dredging. Geophysical surveys must follow the G&G PDCs in Appendix G, and geotechnical surveys, if used to collect sediment samples, are not allowed to penetrate coral hardbottom.

CORAL.7 All equipment with overflow

- Equipment with overflow will be positioned as far from hardbottom as possible and preference will be given to placing overflow equipment in areas where the tides and currents move turbidity away from hardbottom.
- To the extent possible, vessels will be operated in a way to minimize the turbidity plume from overflow through all available methods. These methods may include minimizing air bubbles through adjustment of the “green valve” in hopper dredges, limiting overflow to times when the vessel and currents are moving in the same direction, limiting overflow by not requiring complete filling of the vessel holding area, or other new methods or technologies developed to minimize turbidity.
- Specific requirements for overflow and turbidity are specified by activity in C-BEACH and C-PIPE.

Table 54. Channel and Borrow Area Dredging Scenarios Covered under the 2020 SARBO within the Range of ESA-Listed Corals.
Authorization is based on the distance between the dredging activity and adjacent hardbottom relative to percent fines.

	Presence of Hardbottom	**No Hardbottom** 0-1000 ft				**Hardbottom** • 0-500 ft from Channels • 0-400 ft from Borrow Areas				**Hardbottom** • 500-1000 ft from Channels • 400-1000 ft from Borrow Areas **No Hardbottom** • 0-500 ft from Channels • 0-400 ft from Borrow Areas			
Dredge Type	Percent Fines	0-5%	Time Limit	5-10%	Time Limit	0-5%	Time Limit	5-10%	Time Limit	0-5%	Time Limit	5-10%	Time Limit
Mechanical		●	None	●	None	X	NA	X	NA	X	NA	X	NA
Cutterhead		●	None	●	None	●	< 18 days	●	< 18 days	●	None	●	< 18 days
Hopper w/ no overflow		●	None	●	None	●	< 18 days	X	NA	●	< 18 days	●	< 18 days
Hopper w/ overflow		●	None	●	None	X	NA	X	NA	●	< 18 days	X	NA
Bed Leveling		●	None	●	None	●	< 18 days	X	NA	●	< 18 days	●	< 18 days
Water Injection		X	NA	X	NA	X	NA	X	NA	X	NA	X	NA
Support vessel w/ overflow		●	None	●	None	X	NA	X	NA	X	NA	X	NA

● = Dredge type allowed
X = Dredge Type Not Allowed
NA = Time limit not applicable

2.3 Beach Nourishment

The following PDCs apply to all projects within the range of ESA-listed corals that include beach nourishment. These PDCs assume that the material to be placed on the beach is less than 10% fines.[88] Placement activities covered under SARBO within the range of ESA-listed corals is limited to beach nourishment (e.g., nearshore placement, side-cast dredging, and ODMDS placement are not covered).

C-BEACH.1 Sand placement for beach nourishment projects will be limited to the previously authorized/permitted and constructed beach fill template (defined as the area where sand is placed between the existing mean high water line waterward to the previously approved and constructed ETOF, as shown in Figure 52. Beach fill templates are provided in SARBA Appendix B for previously authorized projects constructed by the USACE Civil Works. Other beach nourishment evaluated and constructed under an individual Section 7 consultation can also be nourished under this Opinion to the previously permitted and constructed beach template.

- If the entire limits of the previously authorized/permitted beach fill template has not been constructed, this Opinion does not cover projects that place sand on coral hardbottom in areas not previously constructed.
- If coral hardbottom occurs within the previously authorized/permitted and constructed beach fill template (i.e., areas where sand has been previously placed/constructed), hardbottom in this area is not considered as functioning *Acropora* critical habitat or "coral hardbottom" as defined in Coral PDCs Section 1.2 for projects outside of the range of *Acropora* critical habitat within the range of ESA-listed corals. Beach sand placement on coral hardbottom in this area is covered under the 2020 SARBO.
- New beach nourishment projects (those not described in the SARBA Appendix B or those without an individual Section 7 consultation that analyzed the effects to ESA-listed *corals* and *Acropora* critical habitat features) within the defined range of ESA-listed corals are not covered under this Opinion.

- Beach nourishment projects in the U.S. Caribbean are not covered under the 2020 SARBO.

[88] Note that this also meets the state of Florida's definition of beach quality sand under Florida Administrative Code Chapter 62B-41.007(2)(j) and Chapter 62B-41.007(2)(k), which provide limitations of the percent fines placed based on the location the material is acquired.

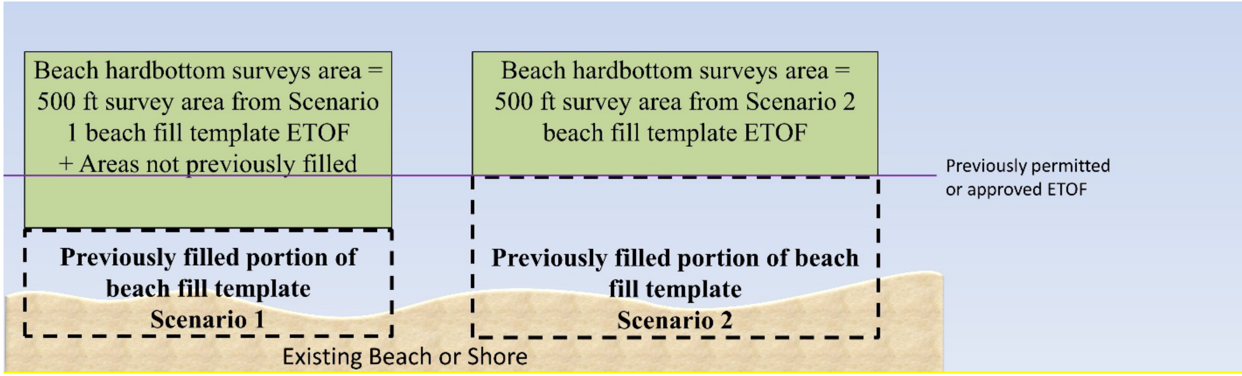


Figure 52. Illustration showing the areas described in the beach nourishment PDCs (Green box represents survey area and dotted line box represents area previously nourished)

C-BEACH.2 Hardbottom surveys will be completed within 2 years prior to beach sand placement for beach nourishment projects within the range of ESA-listed corals (range defined in Coral PDC Section 1.2.1). The surveys areas (referred to as the beach hardbottom survey area) are depicted in using 2 scenarios to describe the survey area.

- If the initial hardbottom survey was conducted using a geophysical survey, the areas identified as hardbottom will be verified using diver surveys, as described in CORAL PDC Section 3. Geophysical surveys must follow the G&G PDCs in Appendix G.
- Beach hardbottom survey area (shown in green in above) will be completed to identify and map the location of any hardbottom located 500 ft waterward of the beach fill template ETOF.
- If the beach fill template includes areas previously permitted/ authorized areas that were NOT previously filled (shown as beach fill template scenario 1 in Figure 52), hardbottom surveys will also be completed in that area of the fill template not previously filled. Areas previously permitted/authorized <u>and</u> previously filled do not require hardbottom surveys within the fill template (scenario 2). Placement of sand on hardbottom and coral within the previously filled beach template is covered under this Opinion.
- If coral hardbottom is NOT identified within the beach hardbottom survey area, then placement of beach quality sand can proceed without additional surveys or monitoring.
- If coral hardbottom IS identified within the beach hardbottom survey area, then all coral hardbottom and ESA-listed corals will be mapped and recorded, as described in CORAL PDC Section 3, and the USACE will contact NMFS for a project-specific review to determine whether coral relocation is appropriate based on anticipated impacts to the identified corals according to the specific site condition through the process outlined in the 2020 SARBO Section 2.9. Conditions that may be considered when evaluating if corals need to be relocated include the composition of sand that

will be placed, hydrographic conditions, proximity to coral, and past experience with similar projects in the area.

C-BEACH.4 Beach nourishment projects will minimize turbidity to ensure that sedimentation does not result in burial of coral or hardbottom outside of the ETOF. Turbidity may be minimized using methods such as the construction of a shore parallel dike in beach areas where sand is hydraulically pumped onto the beach to allow settling of sand prior to discharge of the return water back into the ocean.

C-BEACH.5 If surveys and reports are required by Florida Department of Environmental Protection for beach nourishment projects, all reports provided to Florida Department of Environmental Protection as part of biological monitoring plans will be submitted to NMFS. If the surveys indicate damage or sediment burial of ESA-listed corals or coral hardbottom outside of the ETOF, then NMFS will use the available information provided in the reports to calculate any estimated impact to *Acropora* critical habitat essential features and/or take of ESA-listed corals to determine if the effects exceed the effects analyzed in the 2020 SARBO.

2.4 Pipelines Requirements

The following PDCs apply to all projects within the range of ESA-listed corals that include the placement of floating or submerged pipelines.

C-PIPE.1 Only existing pipeline corridors provided in SARBA Appendix B are covered under this Opinion. No pipeline corridors were identified in the U.S. Caribbean.

C-PIPE.1 All pipelines (anchored or floating) will be placed in a 25-ft-wide pipeline corridor that is selected to minimize and avoid placing the pipeline on coral hardbottom to the maximum extent practicable. Beach nourishment pipeline corridors are typically pre-defined and reused for each nourishment event to minimize additional impacts.

C-PIPE.2 All pipelines will be of sufficient size or weight to prevent movement outside the 25-ft-wide pipeline corridor. Additional anchoring may be needed to achieve this requirement. Floating pipeline or risers will be used when pipelines cross coral hardbottom.

C-PIPE.3 Pipeline Pre-Construction Surveys

- Hardbottom survey area: Hardbottom will be identified within the 25-ft wide pipeline placement corridor and within 100 ft of both sides of it for a total of a 225-ft wide pipeline survey area. If the initial survey is a geophysical survey, the areas identified as hardbottom will be verified using diver surveys, as described in CORAL PDC Section 4.1.
- If coral hardbottom is identified within the 225-ft wide pipeline survey area:
  - A diver survey will be conducted to map the extent of coral hardbottom within the 225-ft wide pipeline survey area and to document all ESA-listed corals within the 25-ft wide pipeline placement corridor, according to the pipeline pre-construction survey protocol outlined in Coral PDC Section 4.1.
  - All ESA-listed corals within the corridor that cannot be avoided (i.e. those within the pipeline footprint whose physical location will result in direct impact of the coral) will be relocated according to the coral relocation protocol outlined in Coral PDC Section 5.

C-PIPE.4 Pipeline During-Construction Surveys
If coral hardbottom is identified within the 225-ft wide pipeline survey area, then additional pipeline during-construction surveys (outlined in Coral PDC Section 4.2) will be required for the length of time that the pipeline is in place. Divers will swim along both sides of the pipe in all areas where the pipe crosses coral hardbottom to determine if there is movement of the pipeline and /or discharge of slurry anywhere along the length of the pipeline. The pipeline during-construction surveys will monitor for the movement of submerged pipelines and support structures for floating pipelines placed near or over hardbottom and to monitor for a discharge of slurry/leaks anywhere along the length of a submerged pipeline near hardbottom or floating pipeline placed over hardbottom. The

pipeline during-construction surveys will be conducted within 24 hours after the pipeline is activated with sand pumping through it, and surveys will continue twice per week until the pipeline is removed, weather and sea conditions permitting.

C-PIPE.5 If a pipeline leak is observed during the pipeline during-construction survey or by the dredging/ pumping crew, the following actions are required:

- Turbidity measurements will be immediately taken at the source of leak (e.g., pipeline / pump station leak site). Substantial leaks are those that result in a turbidity reading that exceeds 29 nephelometric turbidity units the leak site.
- All dredging / pumping / filling operations will cease immediately if a substantial leak is found.
- All dredging / pumping / filling operations will also cease immediately if impacts to coral hardbottom resources are observed, such as sediment accumulation on coral hardbottom and/or physical damage to ESA-listed corals.
- NMFS staff will be notified within 24 hours of documented / observed substantial leaks resulting in turbidity, sedimentation accumulation, or physical impacts to coral hardbottom.
- Dredging / pumping/ filling operations can resume once corrective action has been verified to stop the leak or correct the cause of physical damage.

C-PIPE.6 If movement of the pipeline is observed (in the course of the pipeline during-construction surveys or by the dredging/pumping crew), then the pipeline will be secured in a manner that significantly reduces movement (e.g., anchoring in areas uncolonized by ESA-listed corals along the pipeline or floating collars).

C-PIPE.7 Pipeline Post-Construction Surveys
Following completion of dredging activities and pipeline demobilization, the following actions are required:

- After the pipeline is removed, the entire length of the pipeline will be visually surveyed for damage using the pipeline post-construction survey methods outlined in Coral PDC Section 4.3.
- If a pipeline leaks and/or physical impacts to coral hardbottom or ESA-listed corals have occurred, then a detailed pipeline impact assessment survey is required to document the extent of the impact as outlined in Coral PDC Section 4.4.
- All post construction reports will be provided to NMFS 60 days following the removal of the pipeline in a digital format as defined in Section 2.9 of the 2020 SARBO.

C-PIPE.8 If the pipeline post-construction survey (Coral PDC C-PIPE.7) indicates physical damage or sediment burial of ESA-listed corals or coral hardbottom from the pipeline, then NMFS will use the available information provided in the pipeline surveys to calculate the estimated impact to *Acropora* critical habitat essential features and/or take of ESA-listed corals to determine if the effects exceed the effects analyzed in the 2020 SARBO.

3 Beach Nourishment Survey Protocol

3.1 Survey Objectives

The objectives of the beach nourishment survey protocol are to identify and map the location of all coral hardbottom and ESA-listed corals located (1) between the proposed beach fill template ETOF and 500 ft waterward of the ETOF and (2) within portions of beach fill templates permitted but previously unfilled for beach nourishment projects covered under the 2020 SARBO (these areas are referred to as the beach hardbottom survey area). If ESA-listed corals are identified in the beach hardbottom survey area, the USACE will coordinate with NMFS to conduct a project-specific review to determine if coral relocation is necessary to protect corals from potential turbidity and sedimentation resulting from the beach nourishment. Conditions that may be considered when evaluating if corals need to be relocated include the composition of sand that will be placed, hydrology, proximity to coral, and past experience with similar projects in the area.

3.2 Surveys for Beach Nourishment Projects

For beach nourishment projects covered under this Opinion, the location of hardbottom may be identified using high-resolution geophysical surveys and will then be visually verified by divers. Divers will swim all areas of hardbottom and map the extent of all hardbottom areas within the beach hardbottom survey area described in Coral PDCs Section 2.3. Hardbottom in the survey area will be identified and also documented if the hardbottom meets the definition of coral hardbottom, defined in Coral PDC Section 1.2.2.

Divers will also identify and record the presence of all ESA-listed corals within the beach hardbottom survey area, according to the *ESA-Listed Coral Colony and Acropora Critical Habitat Survey Protocol*, Updated July 2019 (https://www.fisheries.noaa.gov/southeast/consultations/regulations-policies-and-guidance). The protocol provides information on staff qualifications, QA/QC procedures, delineating *Acropora* critical habitat features, coral survey protocols, and data collection requirements. If this guidance is updated, the new NMFS survey protocol will be followed.

3.3 Survey Reports for Beach Nourishment Projects

Surveys will report the information listed below to NMFS within 60 days of the completion of the survey. This information will be collected and reported as described in the 2020 SARBO Section 2.9. The *ESA-Listed Coral Colony and Acropora Critical Habitat Survey Protocol* does not provide a reporting form for surveys associated with beach nourishment projects, but the forms in the protocol can be adapted to this survey type. If this guidance is updated, the new NMFS survey protocol will be followed. The information reported will include:

1. Georeferenced map (ArcGIS files) and GPS coordinates for all hardbottom and ESA-listed corals identified by species.
2. Map of the location of each colony of ESA-listed corals.

3. Map of the location of *Acropora* critical habitat essential feature (i.e. coral hardbottom). Mapping the location of coral hardbottom both within the geographic boundaries of *Acropora* critical habitat and within the range of ESA-listed corals is required, but indicate the area of coral hardbottom that is within Acropora critical habitat.
4. Dimensions of the colony (length, width, and height, or longest dimension length [units = cm]), percent live tissue, and recent partial mortality.
5. Water depth and general description of the vertical relief (high, medium, low) of the coral hardbottom feature where the colony is found.
6. Report summarizing field-data collection.

4 Pipeline Survey Protocol

The following protocols apply to the PDCs required when a pipeline is placed within the range of ESA-listed corals, as defined Coral PDC Section 1.2 above.

4.1 Pipeline Pre-Construction Survey

If coral hardbottom is identified by the geophysical surveys within the 225-ft wide pipeline survey area (25-ft wide pipeline placement corridor and within 100 ft of both sides of it), then the area will be visually surveyed by divers.

- Divers will swim all of the 225-ft wide pipeline survey area where the pipeline will cross coral hardbottom.
- Divers will swim side-by-side, from offshore to inshore, at a distance of 1 m above the surface and will photograph any coral hardbottom that occurs within the proposed pipeline footprint for comparison in the post-construction survey. Photos will be taken from approximately 1 m above the surface and will be focused straight down. A meter stick will be included in the photo for scale. Photos will be numbered and corresponding coral hardbottom patches on the habitat maps will be noted.
- All ESA-listed corals visible within the 25-ft wide pipeline placement corridor will be identified, and any that cannot be avoided (i.e. those within the pipeline footprint whose physical location will result in a direct impact of the coral) will be relocated according to the coral relocation protocol (Coral PDC Section 5). ESA-listed corals within the 25-ft wide pipeline placement corridor that will not be relocated (i.e. those not within the physical pipeline footprint) will be recorded (species name, maximum dimension, and location) and photographed for post-construction comparison.

4.2 Pipeline During-Construction Corridor Survey

If coral hardbottom is identified within the 225-ft wide pipeline survey area, then pipeline during-construction coral surveys are required.

- Diver surveys will start immediately (within 24 hours) following pipeline placement, weather and sea conditions permitting.

- Divers will swim along both sides of the pipe in all areas where the pipe crosses coral hardbottom to determine if there is movement of the pipeline and /or discharge of slurry anywhere along the length of the pipeline. In the event that movement or discharge/slurry is discovered, the measures described in C-PIPE.5 will be followed.
- Diver will inspect the pipe twice per week, weather and sea conditions permitting, until the pipeline is removed.

4.3 Pipeline Post-Construction Survey

A post-construction diver visual inspection will be conducted following construction and after the pipeline is removed.

- After the pipeline is removed, divers will survey the 25-ft wide pipeline placement corridor in the areas where the pipeline crossed coral hardbottom.
- Divers, working in teams of 2, will swim side-by-side at a distance of 1 m above the surface and will photograph any coral hardbottom that occurs within the 25-ft wide pipeline placement corridor. Photos will be taken from approximately 1 m above the surface and will be focused straight down. A meter stick will be included in the photo for scale. Photos will be numbered, and corresponding coral hardbottom patches on the habitat maps will be noted.
- Comparisons will be made between the pre- and post-construction photographs, and any damage to ESA-listed coral or designated critical habitat will be reported to NMFS within 30 days. Reports will indicate if the damage is believed to be unrelated to the project and the reason for the determination.

4.4 Pipeline Impact Assessment Survey

If pipeline leaks or physical impacts (damage or burial) to coral hardbottom or ESA-listed corals have occurred, then a detailed quantitative impact assessment is required per Coral PDC C-Pipe 8.

- Divers, working in teams of 2, will visually survey any area where a leak has been detected or physical damage to coral hardbottom has been recorded during any of the pipeline surveys above.
- Impact assessments will include a delineation (using GPS) of all areas in which coral hardbottom has been damaged, injured, buried, or stressed and will extend out to the furthest extent of such damage, even if the damage extends beyond 225-ft wide pipeline survey area.
- The condition of impacted benthic organisms will be assessed, photographed, and documented.

A pipeline impact assessment survey form has not been developed, but can be completed in coordination with NMFS. At a minimum, the following information will be collected, recorded, and submitted in a digital spreadsheet according to the guidelines in 2020 SARBO Section 2.9.3.1:

- Species name of all ESA-listed corals that have been impacted;
- Dimensions of any impacted colony including the diameter or longest dimension (units = cm);

- Percent live tissue and recent percent mortality (recorded in 10% increments);
- Photograph: Photos will be taken from a position directly above the coral from a distance that allows the entire colony to be in the frame, and a ruler will also be included in the photo for scale. For corals exhibiting signs of sediment stress, close-up photographs will be taken to document stress;
- Sediment cover: Any dusting or accumulation of sediments and all signs of sediment stress will be reported, including the presence of a sediment halo (or partial mortality typically around the base of the colony), the presence of sediment or partial mortality in concave areas of encrusting and massive shaped colonies, and the presence of sediment or partial mortality on the upslope side of colonies growing on steep surfaces;
- GPS coordinates of each impacted colony;
- Site map with locations of each colony and each area of coral hardbottom impacted;

4.5 Pipeline Coral Survey Reports

Results of pipeline coral surveys listed below will be reported to NMFS as described in the bullets below and according to the reporting requirements outlined in the 2020 SARBO Section 2.9.

- Pre-construction pipeline corridor survey: Reported within 10 days of survey completion.
- During-construction pipeline corridor survey: Reported to NMFS within 24 hours if a pipeline leak or impacts to coral hardbottom or ESA-listed corals are detected. All during-construction survey reports will be submitted with the post-construction report.
- Post-construction pipeline corridor survey: Reported to NMFS within 60 days of the removal of the pipeline along with the during-construction reports.
- Pipeline impact assessment survey: Reported to NMFS within 30 days of completion of the survey.

All pipeline coral survey reports will include (1) the data sheets used during the survey (no specific format is required), (2) the photographs collected during the impact assessment, and (3) the GPS coordinates of the location(s) of any impacted coral hardbottom and/or ESA-listed coral. GIS mapping results for areas with impacted resources will also be provided, as a collection of shapefiles (ArcGIS files). For shapefiles, polygons will represent the *in situ* delineated edge of each area containing impacted resources. The specific data that will be collected is provided for each survey type in this section.

5 Coral Relocation Protocol for ESA-Listed Corals

All coral relocation completed for beach nourishment or pipeline placement projects covered under the 2020 SARBO will be completed as described below.

The USACE may contact NMFS prior to a coral relocation project (from either a beach nourishment or pipeline placement project) to determine, through a project specific review, if it may be appropriate to give relocated ESA-listed corals to a coral nursery instead of relocated to a nearby location. If corals are provided to a coral nursery, no monitoring of transplant success (Coral PDC Section 5.4) is required.

For beach nourishment projects, the USACE will contact NMFS prior to relocating corals located between the proposed beach fill template ETOF and 500 ft waterward of the ETOF and in areas of the permitted beach fill template that have not been previously filled, to determine if relocation is necessary based on the likelihood of turbidity or sedimentation reaching corals within this area. This assessment will consider the material to be placed, site conditions, hydrology, and likelihood of potential burial of corals in the area during or after sand placement.

5.1 Qualified person

All relocation and reporting activities will be conducted by staff that meet the requirements outlined in the *ESA-Listed Coral Colony and Acropora Critical Habitat Survey Protocol*, Updated July 2019 (https://www.fisheries.noaa.gov/southeast/consultations/regulations-policies-and-guidance). If this guidance is updated, the new NMFS survey protocol will be followed.

5.2 Relocation site selection

All relocation of ESA-listed coral will be to suitable habitat:

- Relocation sites will occur near the coral's original location, but not within 1,000 ft of the pipeline, dredging footprint, or sand placement area. Relocated corals will be placed in water depths from the mean high water line to 30 m (98 ft) and be within a similar depth as the origin coral location (+/- 5 ft).
- Relocation sites must consist of coral hardbottom or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover.
- Relocation sites will have appropriate water quality (based on water quality data and local knowledge) and minimal chances of other disturbances (future coastal construction, boat groundings, damage caused by curious divers/fisherman).

5.3 Relocation techniques

All colonies will be collected carefully using a hammer and chisel. Upon collection, the colonies will be kept at the original depth until transplantation commences (i.e., cached on site). Transplantation will occur as soon as operationally feasible, but no more than 24 hours after the colony is removed from its original location. During transportation to the transplant site, the corals will be kept in seawater at all times, covered with a lid or towel during transport, and maintained at a water temperature within 2 degrees of ambient water temperature. Transplanted colonies will be placed no closer than 0.75 m from each another.

5.4 Monitoring of Transplanted Corals

Depending on the numbers of relocated corals, all or a subset of those corals will be monitored to determine the success of transplanting. If large numbers of corals are relocated, a subset of colonies representing an appropriate cross section of the species and size classes will be monitored. If the number of corals relocated are 100 or more, the USACE will use power analysis on the total number of relocated corals to determine an appropriate subset of corals to be monitored. The subset will be sufficient to detect a 10% change. The subset will not be less than 20% of the total. The subset will be selected randomly across sites to be representative of the relocated corals. All transplanted corals will be monitored using the methods listed below. Transplanted colonies will be monitored at the time of the transplantation (baseline) and at 5 post-transplant monitoring events. Monitoring requirements here are intended to align with the Florida Fish and Wildlife Conservation Commission coral relocation monitoring guidelines. At the time this Opinion was issued the monitoring guidelines were not yet posted to their website. We will include the link on SERO's Dredge webpage (https://www.fisheries.noaa.gov/content/southeast-dredging) once they do. Reports documenting the transplantation of corals will be submitted to NMFS as required by the PDCs, including the project specific information, and reporting information outlined in the 2020 SARBO Section 2.9. In addition, the transplantation information listed below will be reported:

- Baseline Observations at the transplant location
    - Record the species and the number on the plastic identification tag adjacent to each transplanted colony.
    - Record the widest length, width, and height of the coral, percent live tissue, and site depth at mean high water of each colony at both the original location and the transplant location.
    - Record the GPS location (in decimal degrees) or the compass bearing and distance (in feet) from a known fixed point, and photograph each transplanted coral with a scale in the photo.
- Monitor post-transplant success and survival
    - Monitoring should be conducted at 1 week, 1, month, 3 months, 6 months, and 12 month post-relocation. The purpose of the monitoring events are as follows:
    - 1 week monitoring checks for attachment success; immediately reattach any corals that are not firmly attached to the hardbottom.
    - 1 and 3-month monitoring records sediment cover on the colonies (sediment dusting, sediment accumulation, partial burial, burial of the base, burial, or sediment halo if present) and colony condition (bleaching, % live tissue, and presence of disease, fouling, or predation).
    - 6 and 12-month monitoring records colony size, percent live tissue, sediment cover on the colonies, and colony condition.
    - Post-transplant monitoring reports, including photographs, will be submitted to NMFS within 30 days of each monitoring event. Reports will include a table with the information described above and percent mortality (reported in 10% increments) for each of the monitored transplanted corals.

1 Year Post-Transplant Success Criteria for a Specific Project

The success of transplanting corals by project (e.g., corals transplanted for x pipeline project or y beach nourishment project) is met if 85% of all of the ESA-listed corals/coral colonies that are transplanted for that project survive the transplant procedure. Survival of each individual coral or colony transplanted for the project is measured by determining if the individual coral or colony has less than 25% partial mortality of the live tissue. The 1-year survival rate may consider the health of existing corals in the surrounding area, meaning that the survival rate may be adjusted if all corals in the area are effected by an external factor such as coral bleaching or disease. During the 2020 SARBO annual review (2020 SARBO Section 2.9.4), a summary will be provided of all ESA-listed corals transplanted associated with all projects covered under this Opinion.

Coral Transplanting Success Criteria for All Projects Covered under this Opinion

The success of coral transplanting under this Opinion will be tracked as part of the 2020 SARBO annual review (2020 SARBO Section 2.9.4) to ensure at least 85% of all corals transplanted for all projects that occur over a 5-year period survive based on the reports for each individual project that transplanted corals. This timeframe was selected to allow time for multiple projects to be completed and monitored for 1 year to determine the success of transplanting corals covered under this Programmatic Opinion. If this 5-year transplanting success metric is not met, the USACE has the option to either reinitiate consultation to consider the effects of the additional loss of corals not considered in this Opinion or to outplant corals of the same species of corals that did not meet the success criteria. Outplanting is the process of moving corals grown in a coral nursery to the relocation site where corals were transplanted. If trained staff perform the coral transplanting, it is expected that the success criteria rate will be met based on monitoring results from similar past projects. If outplanting is chosen, the number of corals transplanted is determined according to the multipliers listed in Table 55 to replace a similar amount of live coral tissue and assure success of the second transplanting. Monitoring success of the second transplanting is completed in the same way as the first transplanting event.

Table 55. Outplanting Ratio if the Coral Relocation Survival Rate was not Met

Minimum outplant sizes are 15 cm for Acropora, 10 cm for Dendrogyra, and 2.5 cm for Orbicella.

Coral Size (cm)	Multiplier for Acropora Corals (i.e., elkhorn and staghorn)	Multiplier for Orbicella Corals (boulder star, mountainous star, lobed star)	Multiplier for Dendrogyra Corals (Pillar)
1-20	1	5	1
21-30	2	10	2
31-40	3	15	3
41-50	4	21	4
51-60	5	27	5
61-70	5	33	6
71-80	6	40	7
81-90	7	46	8
91-100	8	53	9
101-110	9	60	11
111-120	10	68	12

Appendix D. 2020 SARBO Johnson's Seagrass PDCs

The PDCs in Appendix D apply to all projects that occur within the range of Johnson's seagrass, as defined in this appendix. These requirements are in addition to any other applicable 2020 SARBO PDCs and the requirements outlined in the 2020 SARBO.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but have substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the of the 2020 SARBO*, will require individual Section 7 consultation. Any area previously authorized or permitted to be dredged or have material placed within the action area and analyzed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.*

1 Description of the Area Where Johnson's seagrass PDCs Apply

Johnson's seagrass PDC Section 1.1 provides information on Johnson's seagrass designated critical habitat. Johnson's seagrass PDC Section 1.2 provides the geographic range of Johnson's seagrass defined for the 2020 SARBO and the area in which adherence to the Johnson's seagrass PDCs is required by the 2020 SARBO.

1.1 Johnson's Seagrass Critical Habitat

According to the Final Rule designating Johnson's seagrass critical habitat (65 FR 17786, Publication Date April 5, 2000), the physical and biological features of the critical habitat areas include: adequate water quality, salinity levels, water transparency, and stable, unconsolidated sediments that are free from physical disturbance. Johnson's seagrass critical habitat limits are defined in the Final Rule, provided in Figure 53 below, and available in GIS format on the NMFS Southeast Region website at https://sero.nmfs.noaa.gov/maps_gis_data/index.html. All 4 features must be present for an area to be considered functioning critical habitat.

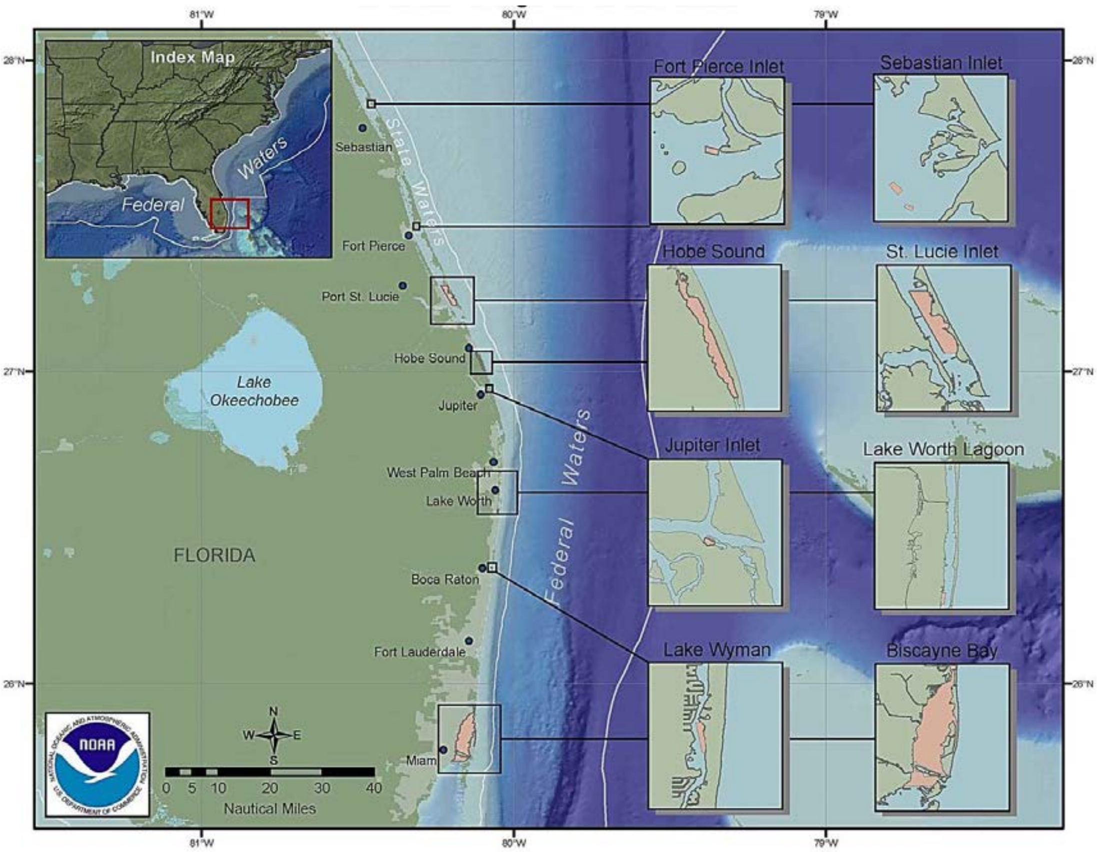


Figure 53. Johnson's seagrass critical habitat units

1.1.1 Areas Omitted from Johnson's seagrass Critical Habitat

The following areas are not considered Johnson's seagrass critical habitat, pursuant to the Final Rule:

a. Areas within the geographic boundary of critical habitat that do not support all 4 physical and biological features listed above. This is the only designated critical habitat within the SARBO action area that requires the presence of all of the physical and biological features to be functioning critical habitat.

b. The 18.5–km-long portion of the navigation channel within the Intracoastal Waterway that occurs within the geographical limits of Johnson's seagrass is excluded from critical habitat (65 FR 17786, Publication Date April 5, 2000, page 17791, item 4)

c. A portion of northern Biscayne Bay. The geographic limits of Johnson's seagrass critical habitat align with the boundaries of the Biscayne Bay Aquatic Preserve. The USACE and NMFS have interpreted this to mean that some of the man-made canals are not in Johnson's seagrass critical habitat because they are not part of the Biscayne Bay Aquatic Preserve. A GIS layer is available as a tool to clarify whether a particular project is located inside or outside of critical habitat based on the Aquatic Preserve geographic boundary (https://sero.nmfs.noaa.gov/maps_gis_data/index.html).

1.2 Areas and terms defined for the 2020 SARBO

1.2.1 Definition of the range of Johnson's seagrass

For the purposes of 2020 SARBO, the range of Johnson's seagrass is defined as the area within the lagoon systems on the east coast of Florida between latitude 28.0328ºN (by the mouth of Turkey Creek/Palm Bay, Palm Bay, Brevard County) south to latitude 25.7500ºN (central Biscayne Bay). While the range of Johnson's seagrass includes the area designated as Johnson's seagrass critical habitat, the Johnson's seagrass PDCs encompass a larger area in order to be protective of the entire range where Johnson's seagrass may be present based on the best available scientific information.

1.2.2 Habitat features necessary to support Johnson's seagrass

As stated in the Final Rule designating Johnson's seagrass critical habitat (65 FR 17786, Publication Date April 5, 2000), this species requires adequate water quality, salinity levels, water transparency, and stable, unconsolidated sediments that are free from physical disturbance to grow and survive. For purposes of the Johnson's Seagrass PDCs, we use the presence of these features throughout the defined range of Johnson's seagrass (which includes, but is not limited to Johnson's seagrass critical habitat) to evaluate locations where Johnson's seagrass may be found. If all 4 of the features necessary to support seagrass are not present at a proposed project site within the defined range of Johnson's seagrass, seagrass surveys are not required. For 2020 SARBO, the following areas are considered to not support the necessary habitat features:

a. Areas deeper than -13 ft are assumed to be too deep to support adequate water transparency for Johnson's seagrass growth. Studies show that Johnson's seagrass occurs in waters shallower than -10-13 ft (-3-4 m) (NMFS 2007a). Water depths greater than -13 ft are not believed to provide the water transparency necessary for sufficient sunlight to reach the sea floor to support Johnson's seagrass growth and are therefore not considered to have the habitat features necessary to support Johnson's seagrass, for the analysis in 2020 SARBO.
b. Consolidated sediments (e.g., hardbottom).
c. Areas with accumulated material that precludes seagrass growth (e.g., thick muck).
d. Areas with poor water quality that do not support any plant growth.

1.3 Johnson's Seagrass Project Action Area

The 2020 SARBO action area is described in the 2020 SARBO Section 2.8. However, the project-specific action area for a specific project is limited to the area where effects from the action occur and is therefore much smaller. For the purpose of the Johnson's Seagrass PDCs, the Johnson's seagrass project action area is defined as including both the project footprint plus the Johnson's seagrass survey areas defined below.

1.3.1 Johnson's Seagrass Project Footprint

For purposes of the Johnson's Seagrass PDCs, the "Johnson's seagrass project footprint" is the area defined by the physical location of the federal action. The project footprint includes all of the areas where activities are occurring related to the project such as the location of dredging or material placement and the location of equipment placement (e.g., pipeline installation, anchoring, and staging areas), and all vessel routes.

1.3.2 Johnson's Seagrass Survey Area

The "Johnson's seagrass survey area" as used in the Johnson's seagrass PDCs refers to the area in which surveys surrounding the project are required. The size and location of the Johnson's seagrass survey area required for a project is provided in the Johnson's seagrass PDCs.

2 Johnson's seagrass PDCs for SARBO

The following PDCs apply to all projects that occur within the range of Johnson's seagrass, as defined in Johnson's seagrass PDC Section 1.2.1 above. These PDCs are in addition to any other applicable PDCs provided in the 2020 SARBO.

2.1 General PDCs and design considerations

JSG.1 Minor Channel Modifications/Realignment is prohibited in areas where Johnson's seagrass is present. Before a modification or realignment is completed within the range of Johnson's seagrass, a visual seagrass survey (conducted and reported according to the requirements in Johnson's Seagrass PDC Section 3) is required to confirm Johnson's seagrass is not present.

JSG.2 Borrow areas within the defined range of Johnson's seagrass in areas less than 13 ft deep are not included in 2020 SARBO, with the exception of the approved Baker's Haulover borrow site shown in Figure 54 with the approximate center of the borrow site located at 25.7478ºN, -80.128ºW(North American Datum 1983).

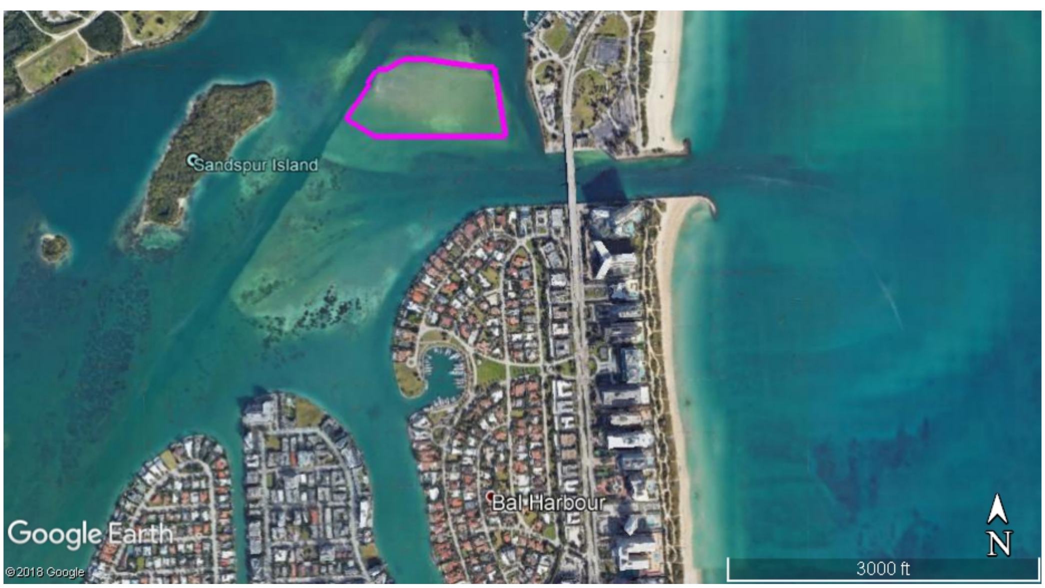


Figure 54. Image of the Baker's Haulover borrow site location provided by the USACE

JSG.3 Geotechnical surveying is prohibited in areas where Johnson's seagrass is present and where the features that support Johnson's seagrass are present (defined in Johnson's Seagrass PDC Section 1.2). Before a geotechnical survey is initiated within the range of Johnson's seagrass, a geophysical survey or visual survey may be used to evaluate the potential presence of seagrass or these features, which will then be confirmed by a visual in-water survey.

JSG.4 Placement of equipment and materials will avoid areas with any seagrasses including turbidity curtain anchors, barge spudding or anchoring, pipelines, or other materials. In cases where pipeline placement cannot avoid seagrass, floating pipelines will be used instead of anchored pipelines.

Johnson's seagrass pre-construction surveys

JSG.5 Within the project footprint (defined in Johnson's seagrass PDC Section 1.3 above) Johnson's seagrass pre-construction survey requirements:
Within the range of Johnson's seagrass, a pre-construction seagrass survey must be completed to determine the presence of Johnson's seagrass within the proposed project footprint to quantify and record Johnson's seagrass that will be removed as a result of the project. Johnson's seagrass pre-construction surveys within a maintained navigation channel can be completed either of the 2 ways listed below:

- Side-scan sonar: A survey must be completed within 1 year prior to the start of dredging. Any seagrass identified using this technique will be assumed to be Johnson's seagrass, unless otherwise determined, and the total area will be counted as loss under 2020 SARBO and recorded according the requirements in the 2020 SARBO Section 2.9f, and/or
- Visual seagrass survey: A visual survey conducted and documented according to the requirements provided in Johnson's Seagrass PDC Section 3 below. A visual survey is not required within the navigation channel if a geophysical survey is completed and all seagrasses observed are assumed to be Johnson's seagrass. Visual surveys in navigation channels may be unsafe to perform due to the amount of vessel traffic in these areas.

JSG.6 Within the project footprint (defined in Johnson's seagrass PDC Section 1.2 above) Johnson's seagrass pre-construction survey requirements:
Within the range of Johnson's seagrass (as defined in Section 1.2 above), a Johnson's seagrass pre-construction survey must be performed in the area extending 500 ft outward from the project footprint along each side of the dredged channel footprint (i.e., within 500 ft from the outer edge of both sides of the channel being dredged on), to determine the presence of Johnson's seagrass, as defined below. All visual seagrass surveys will be conducted and reported according to the requirements provided in Johnson's Seagrass PDC Section 3 below.

- A preliminary reconnaissance seagrass survey may be completed prior to a visual seagrass survey using boat/snorkel reconnaissance, GIS data available that maps the locations of historic seagrass areas, or by side-scan sonar to determine if any seagrasses are present. In areas where seagrasses are identified as present, a visual seagrass survey is required to determine if the seagrass bed composition includes Johnson's seagrass.

- Visual Johnson's seagrass surveys will be performed within all areas of the defined Johnson's seagrass survey area where preliminary surveys (including GIS data) indicate the potential presence of seagrass. If no preliminary survey was completed, the entire Johnson's seagrass survey area will be surveyed according to the requirements of the visual seagrass survey.

2.2 Dredging operations

JSG.7 Within the range of Johnson's seagrass, dredging is limited to methods that minimize turbidity and reduce the likelihood of adjacent seagrass burial:

- Suction dredging will be preferred such as cutterhead or small hopper dredge. Unless overflow from the hopper or associated barges and scows is necessary (see next bullet for overflow requirements), the use of turbidity curtains is not required.
- Overflow of scows, hopper dredges, and barges will be minimized to the maximum extent practicable. If scows, barges, or hopper dredges are located within 500 ft of Johnson's seagrass (based on pre-construction visual seagrass surveys conducted according to the survey and reporting requirements provided in Johnson's Seagrass PDC Section 3 below), turbidity curtains will be installed around the seagrass beds to protect them from turbidity or along the outer edge of the channel if Johnson's or other non-listed seagrasses extend to the channel's edge. Turbidity curtains will not be removed until turbidity subsides to background levels.
- Mechanical dredging: If mechanical dredging (e.g., clamshell or bucket dredge) and bed-leveling is used within the range of Johnson's seagrass, turbidity curtains will be used to protect all seagrasses within 500 ft of the channel edge (based on pre-construction visual seagrass surveys conducted according to the survey and reporting requirements provided in Johnson's Seagrass PDC Section 3 below). The turbidity curtains will be placed either around the seagrass bed to protect it from turbidity or along the outer edge of the channel if seagrasses are identified immediately adjacent to the channels edge.
- Water-injection dredging will not occur within 1,000 ft of Johnson's seagrasses.
- Muck dredging (as defined in the 2020 SARBO Section 2.3.3), will be accomplished by suction dredging methods to minimize turbidity. In areas that meet the definition of muck dredging as defined in 2020 SARBO, seagrasses and Johnson's seagrass critical habitat essential features would not be present.

JSG.8 In-water placement of dredged material

- Within the geographic limits of Johnson's Seagrass critical habitat (as defined in Johnson's seagrass PDC Section 1.1), in-water material placement (e.g., side-cast dredging, nearshore placement) will not occur.
- Within the range of Johnson' seagrass (as defined in Johnson's Seagrass PDC Section 1.2), but outside of its designated critical habitat, in-water material placement may be accomplished if:
  - In waters -13 ft deep or deeper OR
  - In areas less than -13 ft deep that are devoid of habitat features necessary to support Johnson's seagrass (as defined in Johnson's seagrass PDC Section 1.2). Placement will not occur in areas that have known historical or present seagrass beds (ESA-listed or non-listed) if the habitat features are present.

JSG.9 Turbidity monitoring during dredging
Within the range of Johnson's seagrass, turbidity will be monitored along the edge of the dredge footprint and at locations 100 ft and 500 ft outside of the footprint. Results from this monitoring will be reported according to the requirements in the 2020 SARBO Section 2.9.

2.3 Post-construction survey

JSG.10 Post-construction surveys outside of the dredging footprint:
Within the range of Johnson's seagrass, if Johnson's seagrass was identified during the Johnson's seagrass pre-construction survey (required in PDC JSG.6), another visual seagrass survey (according to the same visual seagrass survey protocols and reporting requirements) will be completed in the same survey area within 1-3 months post-construction to determine any loss of Johnson's seagrass resulting from the project.

3 Johnson's seagrass surveys

A visual seagrass survey will be conducted as required by the Johnson's seagrass PDCs to determine the presence Johnson's seagrass and/or to quantify the loss of the ESA-listed species for projects covered under 2020 SARBO. For the purposes of 2020 SARBO, a visual seagrass survey will be conducted within 1 year prior to the start of dredging and preferably during the peak growing season between June 1 and September 30.

Visual seagrass surveys will be conducted according to the current NMFS Johnson's seagrass survey protocol provided in the *Recommendations for Sampling Halophila johnsonii at a Project Site* as provided in Appendix III of the Johnson's Seagrass Recovery Plan (65 FR 17786, Publication Date April 5, 2000) available at: https://www.fisheries.noaa.gov/resource/document/recovery-plan-johnsons-seagrass-halophila-johnsonii. This survey protocol may be updated if new information warrants a revision. The current and any revised versions will also be available at the NMFS SARBO webpage (https://www.fisheries.noaa.gov/content/southeast-dredging) for easy reference.

The results of the visual seagrass survey will document and report the information listed below according to the reporting requirements outlined in 2020 SARBO Section 2.9. A specific data collection table or format is not required so long as the survey details listed below are provided in a format that can be easily viewed and sorted.

- Document the location of Johnson's seagrass in latitude and longitude using decimal degrees (i.e., xx.xxxxºN, –xx.xxxxºW) and include a shapefile of the extent of the area identified with Johnson's seagrass.
- Record the size of the area supporting Johnson's seagrasses in square feet or acres
- Estimate of percent coverage of Johnson's seagrass
- Estimate the density of Johnson's seagrass
- Record the presence of other non-ESA-listed seagrasses by species identified in the area.
- For post-construction surveys required by PDC JSG.12, record the location and size of the area if Johnson's seagrass identified during the pre-construction survey was buried, damaged, or removed.

Appendix E. 2020 SARBO Sturgeon PDCs

The PDCs in Appendix E apply to all projects that occur within rivers where Atlantic and shortnose sturgeon are known to occur, as defined in this appendix. These requirements are in addition to any other applicable PDCs outlined the 2020 SARBO PDCs.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but have substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the 2020 SARBO, will require individual Section 7 consultation. Any area previously authorized or permitted to be dredged or have material placed within the action area and analyzed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

1 Description of the Area Sturgeon PDCs Apply

For the 2020 SARBO, the Sturgeon PDCs were designed to protect all 5 DPSs of Atlantic sturgeon and shortnose sturgeon and critical habitat features for Atlantic sturgeon critical habitat in the action area. Within the action area, there is designated critical habitat for the Carolina and SA DPSs of Atlantic sturgeon. The location of each Atlantic sturgeon critical habitat unit and the features of critical habitat essential to conservation of the Carolina and SA DPSs are provided in the Sturgeon PDC Section 1.1.

Sturgeon PDC Section 1.2 defines the area where these PDCs will apply. This area includes rivers outside of designated critical habitat for Atlantic sturgeon, but where adherence to the sturgeon PDCs is still required for projects covered under 2020 SARBO. Atlantic and shortnose sturgeon have similar life history characteristics, occupy many of the same rivers, and use many of the same general habitat types for similar life cycle functions. Therefore, PDCs protective of one species, will largely be protective of the other.

1.1 Atlantic Sturgeon Critical Habitat and its Features

Fourteen units of Atlantic sturgeon critical habitat occur within action area. This includes 7 units of critical habitat designated for the Carolina distinct population segment (DPS)(i.e., North Carolina to northern portions of South Carolina) and 7 units designated for the SA DPS (i.e., southern portions of South Carolina to the Florida/Georgia border). These critical habitat units are defined in the Final Rule designating Atlantic Sturgeon critical habitat (82 FR 39160; Publication Date August 17, 2017) and provided in Figure 55 below. They are also available for download on the NMFS Southeast Region website at https://sero.nmfs.noaa.gov/maps_gis_data/index.html.

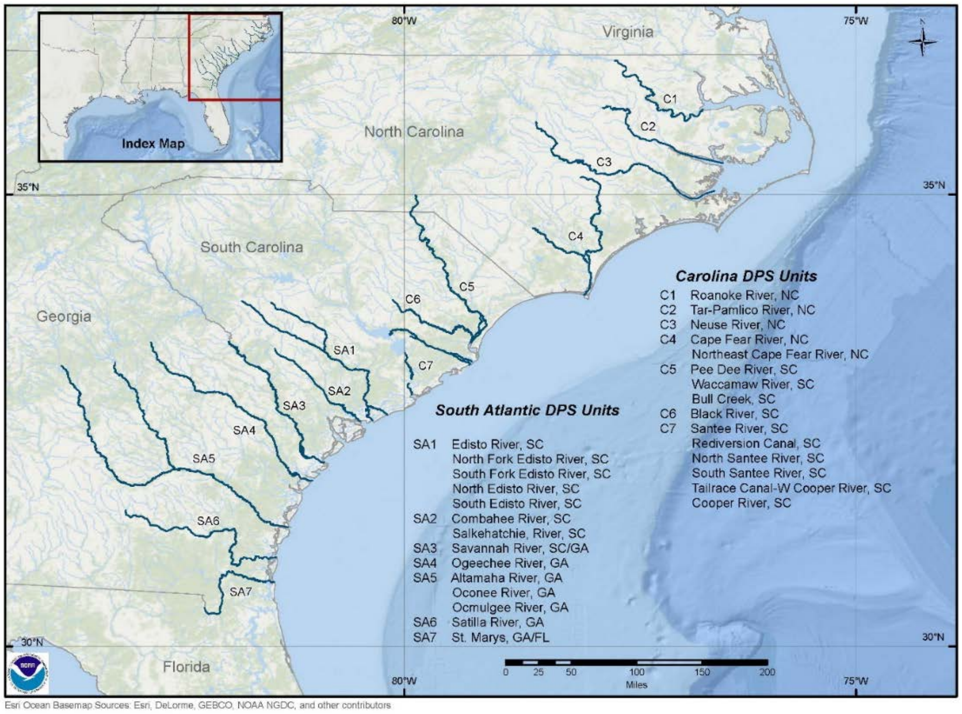


Figure 55. Atlantic sturgeon critical habitat rivers

The Final Rule designating Atlantic sturgeon critical habitat (82 FR 39160; Publication Date August 17, 2017) determined the 4 physical features essential for the conservation of Atlantic sturgeon belonging to the Carolina and SA DPSs are those habitat components that support successful reproduction and recruitment. These are:

(1) Hardbottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0-0.5 parts per thousand range) for settlement of fertilized eggs and refuge, growth, and development of early life stages;

(2) Aquatic habitat inclusive of waters with a gradual downstream gradient of 0.5 up to as high as 30 parts per thousand and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development;

(3) Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support:
- (i) Unimpeded movement of adults to and from spawning sites;
- (ii) Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and
- (iii) Staging, resting, or holding of subadults or spawning condition adults. Water depths in main river channels must also be deep enough (at least 1.2 m) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river;

(4) Water quality conditions, especially in the bottom meter of the water column, with temperature and oxygen values that support:
- (i) Spawning;

(ii) Annual and inter-annual adult, subadult, larval, and juvenile survival; and
(iii) Larval, juvenile, and subadult growth, development, and recruitment. Appropriate temperature and oxygen values will vary interdependently, and depending on salinity in a particular habitat. For example, 6.0 mg/L DO or greater likely supports juvenile rearing habitat, whereas DO less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25°C. In temperatures greater than 26°C, DO greater than 4.3 mg/L is needed to protect survival and growth. Temperatures of 13 to 26 °C likely support spawning habitat.

1.2 Area where Sturgeon PDCs Apply

For 2020 SARBO, the Sturgeon PDCs apply in the rivers used by Atlantic and/or shortnose sturgeon, including those designated as Atlantic sturgeon critical habitat for the Carolina and SA DPSs (82 FR 39160, Publication Date August 17, 2017), and also in smaller rivers in the action area that Atlantic and/or shortnose sturgeon occupy for a portion of the year. All rivers where PDCs apply are listed in Table 56 and in Sturgeon PDCs Section 2 and are referred to as sturgeon rivers throughout the document.

2 Sturgeon PDCs for SARBO

The following PDCs apply to all rivers occupied by Atlantic and/or shortnose sturgeon, as defined in Sturgeon PDC Section 1.2 above. The sturgeon PDCs provided below are in addition to any other applicable PDCs provided in 2020 SARBO.

STURGEON.1 No in-water placement is allowed in sturgeon rivers, including side-cast dredging and nearshore placement.

STURGEON.2 Dredging operations, including related equipment, and projects conducted by other entities in the vicinity will not block more than 50% of the sturgeon river width to allow safe passage of sturgeon.

STURGEON.3 Dredging in sturgeon rivers will follow the restrictions in Sturgeon PDC Section 3. Activities are not covered under this Opinion that occur above the upstream river mile identified in the tables in this section for each sturgeon river.

3 Sturgeon River Dredging Restrictions

Table 56 identifies portions of rivers, and times of year, where dredging may be harmful to Atlantic and/or shortnose sturgeon in all known sturgeon spawning rives as well as to protect the essential features of Atlantic sturgeon critical habitat. Information was gathered to identify all of the known Atlantic and shortnose sturgeon aggregation sites so that those sections of the river could be protected during periods of time that higher numbers of sturgeon may be present. In addition, water quality data in these rivers was reviewed from the available water quality monitoring gauges that are referenced in the tables. Areas of the river were identified that had recorded DO levels and temperatures that may be stressful to sturgeon using the limits of DO and temperature identified in Atlantic sturgeon critical habitat rule for critical habitat physical feature 4 (provided in Sturgeon PDC Section 1.1 above).

Table 56 identifies all sturgeon rivers along with the river miles and months the restrictions apply.[89] The restrictions identified are A-D described below. The restrictions are based on a combination of available river water quality and seasonal aggregation information, which will be updated during the SARBO annual review (Section 2.9 of 2020 SARBO) as with any available new information. Aggregation areas identified in Table 56 are also shown in Figure 56-Figure 72 Projects occurring in these sturgeon spawning rivers must follow the most recent information and restrictions in the tables to be covered under 2020 SARBO, which will be available on the NMFS SARBO webpage (https://www.fisheries.noaa.gov/content/southeast-dredging)

[89]River Mile zero – River mile zero is defined as the intersection of the river middle line with a line running from the two farthest downstream natural river edges (does not include jetties).

Table 56. Sturgeon River Dredging Restrictions

River	Section of River (river mile [RM])	Aggregation Data (river mile)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Chowan River	Any (no upper river work limit)	No data	A	A	A	A	A	**D**	**D**	**D**	**D**	A	A	A
Roanoke River	0-8.8 (near Weyerhaeuser Paper Plant)	No data	A	A	A	A	A	A	**B**	A	A	A	A	A
Roanoke River	8.8-127 (U.S. 301 Bridge, near Weldon, North Carolina) (upper river work limit)	No data	A	A	A	A	A	A	A	A	A	A	A	A
Tar - Pamlico River	0-21.9 (mouth of Bath Creek)	No data	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Tar - Pamlico River	21.9-60 (U.S. Highway 13, near Greenville, North Carolina) (upper river work limit)	No data	A	A	A	A	A	A	A	A	A	A	A	A
Neuse River	0-23.8 (near Neuse River Recreation Area)	No data	A	A	A	**B**	**B**	**B**	**B**	**B**	**B**	**B**	A	A
Neuse River	23.8-37.5 (near Shad Cove/Goose Creek)	No data	A	A	A	A	A	**B**	**B**	**B**	**B**	**B**	A	A
Neuse River	37.5-43 (Route 43 Bridge, near (upper river work limit)	No data	A	A	A	A	A	A	A	A	A	A	A	A
New River	0-5.5 (Sneads Ferry Rd Bridge/Route 172)	No data	A	A	A	A	A	A	A	A	A	A	A	A
New River	5.5+ (no upper river work limit)	No data	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Cape Fear River	0-21.9 (near Watermark Marina)	None identified	A	A	A	A	A	A	**B**	**B**	**B**	A	A	A
Cape Fear River	21.9-40 (upstream of Lake Sutton) (upper river work limit)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Northeast Cape Fear River	0-23.1 (Confluence with Cape Fear River – I-40 Bridge)	No data	A	A	A	A	A	**B**	**B**	**B**	**B**	**B**	A	A
Northeast Cape Fear River	23.1-40 (Croomsbridge Road Bridge) (upper river work limit)	No data	A	A	A	A	A	A	**B**	A	A	A	A	A
Great Pee Dee River	0-6.3 (near Jehrico Creek)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A

River	Section of River (river mile [RM])	Aggregation Data (river mile)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Great Pee Dee River	6.3-29.4 (U.S. 701 Bridge)	None identified	A	A	A	A	A	A	A	**B**	**B**	A	A	A
Great Pee Dee River	29.4-43 (near Petersfield Landing) (upper river work limit)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Waccamaw River	0-8.1 (near Hagley Landing)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Waccamaw River	8.1-21 (near Bucksport Marina) (upper river work limit)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	**B**	A	A
Bull Creek/Big Bull Creek	0-62 (Confluence with Waccamaw River – Confluence with Great Pee Dee River)	No data	A	A	A	A	A	**D**	**D**	**D**	**D**	A	A	A
Black River	0-4.1 (near Wedgefield Plantation Country Club)	None identified	A	A	A	A	A	A	**B**	**B**	**B**	A	A	A
Black River	4.1-44 (Simms Reach Rd Bridge) (upper river work limit)	None Identified to RM 15; No data above RM 15	A	A	A	A	A	**B**	**B**	**B**	A	A	A	A
Sampit River	Any (no upper river work limit)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Rediversion Canal	0-5 (Confluence with Santee Mainstem-Base of St. Stephen Powerhouse)	No data	A	A	A	A	A	**D**	**D**	**D**	**D**	A	A	A
Santee River	Confluence with North and South Santee-64 (U.S. 52 Bridge) (upper river work limit)	None identified	A	A	A	A	A	A	A	A	A	A	A	A
North Santee River	0-Confluence with Santee Mainstem	None identified	A	A	A	A	A	A	A	A	A	A	A	A
South Santee River	0-Confluence with Santee Mainstem	None identified	A	A	A	A	A	A	A	A	A	A	A	A
Cooper River	0-1.9 (U.S. 17 Bridge)	None identified	A	A	A	A	A	A	A	A	A	A	A	A
Cooper River	1.9-10.6 (I-526 Bridge)	None identified	A	A	A	A	A	A	**B**	**B**	**B**	A	A	A
Cooper River	10.6-35 (Railroad trestle bridge) (upper river work limit)	22.5-31(near Bushy Point)	A	A	A	A	A	**C**	**C**	**C**	**C**	**C**	A	A

River	Section of River (river mile [RM])	Aggregation Data (river mile)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Ashley River	0-9.7 (I-526 Bridge)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Ashley River	9.7-22.5	None identified	A	A	A	A	**B**	**B**	**B**	**B**	**B**	A	A	A
Ashley River	22.5+ (no upper river work limit)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
Edisto River	Confluence with North and South Edisto-49 (U.S. Alternate 17 Bridge) (upper river work limit)	None identified	A	A	A	A	A	A	A	A	A	A	A	A
North Edisto River	0-5 (near Leadenwah Creek)	None identified	A	A	A	A	A	A	A	**B**	A	A	A	A
North Edisto River	5-12.5 (Confluence with Edisto River)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A
South Edisto River	0-8.8 (near Raccoon Island)	None identified	A	A	A	A	A	A	**B**	A	**B**	A	A	A
South Edisto River	8.8-12.5 (Confluence with Edisto River)	None identified	A	A	A	A	A	A	A	A	A	A	A	A
Ashepoo River	Any (no upper river work limit)	No data	A	A	A	A	A	**D**	**D**	**D**	**D**	A	A	A
Combahee - Salkehatchie River	0-8.2 (near Fields Point)	No data	A	A	A	A	A	A	**B**	**B**	**B**	A	A	A
Combahee - Salkehatchie River	8.2-60 (I-95 bridge)	No data	A	A	A	A	A	**D**	**D**	**D**	**D**	A	A	A
Savannah River	0-65 (Railroad Avenue/Clyo Road near Clyo, Georgia)	21.1 (near Port Wentworth, Georgia)-32	A	A	A	A	**C**	**C**	**C**	**C**	**C**	**C**	A	A
Ogeechee River	0-62 (State Route 26/U.S. 80 bridge near Eden, Georgia) (upper river work limit)	24-37 (near Richmond Hill, Georgia)	A	A	A	A	A	**C**	**C**	**C**	**C**	**C**	A	A
Altamaha River	0-93 (Confluence Oconee/ Ocmulgee Rivers)	10.5-17.5 (near I-95; U.S. Alternate 17 Bridges)	A	A	A	A	**C**	**C**	**C**	**C**	**C**	**C**	A	A
Satilla River	0-50	20-28.75	A	A	A	A	A	A	**C**	**C**	**C**	**C**	A	A

River	Section of River (river mile [RM])	Aggregation Data (river mile)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
	(Georgia State Route 252 Bridge) (upper river work limit)	(Woodbine, Georgia)												
St. Mary's River	0-4 (Intracoastal Waterway [IWW])	None identified	A	A	A	A	A	A	A	A	A	A	A	A
St. Mary's River	IWW-21 (I-95 bridge) (upper river work limit)	12.5-22	A	A	A	A	**C**	**C**	**C**	**C**	**C**	**C**	A	A
St Johns	0-11 (I-295 bridge near Blount Island, Florida)	None identified	A	A	A	A	A	A	A	**B**	**B**	A	A	A
St Johns	11+ (no upper river limit)	None identified	A	A	A	A	A	**B**	**B**	**B**	**B**	A	A	A

The dredging restriction categories for sturgeon rivers described below apply only to Sturgeon Rivers in the sections and times identified in Table 56. The decision process of how restrictions about restrictions were made is based on the information available as described in Table 57. Sturgeon River Dredging Restriction Decision Process Based on Available Information. These restrictions are in addition to the 2020 SARBO PDCs such as hopper dredging requirements and reporting requirements.

A. All dredge types may be used during these months.

B. All dredge types may be used during these months. However, cutterhead dredging will be monitored for take according to the guidelines in Sturgeon PDC Section 4.

C. All dredge types may be used during these months outside of the aggregation areas identified by time and location in Table 56. Aggregation areas are also shown in Figure 56-Figure 72. Cutterhead dredging will be monitored for take according to the guidelines in Sturgeon PDC Section 4.

D. No dredging allowed during these months

Table 57. Sturgeon River Dredging Restriction Decision Process Based on Available Information

Aggregation Area	Water Quality Data	Sturgeon River Dredging Restriction
Data available: None identified	Data available: Water quality acceptable	A
Data available: None identified	Data available: Historically low DO and high temperatures	B
Data available: None identified	No data available	B (June 1–September 30)
Data available: Areas identified	Data available: Water quality acceptable	C
Data available: Areas identified	Data available: Historically low DO and high temperatures	C
Data available: Areas identified	No data available	C
No data available	Data available: Water quality acceptable	A
No data available	Data available: Historically low DO and high temperatures	B (June 1–September 30)
No data available	No data available	D (June 1–September 30)

Figure 56. Roanoke River
River Mile 0 is marked with a yellow pin and yellow line. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

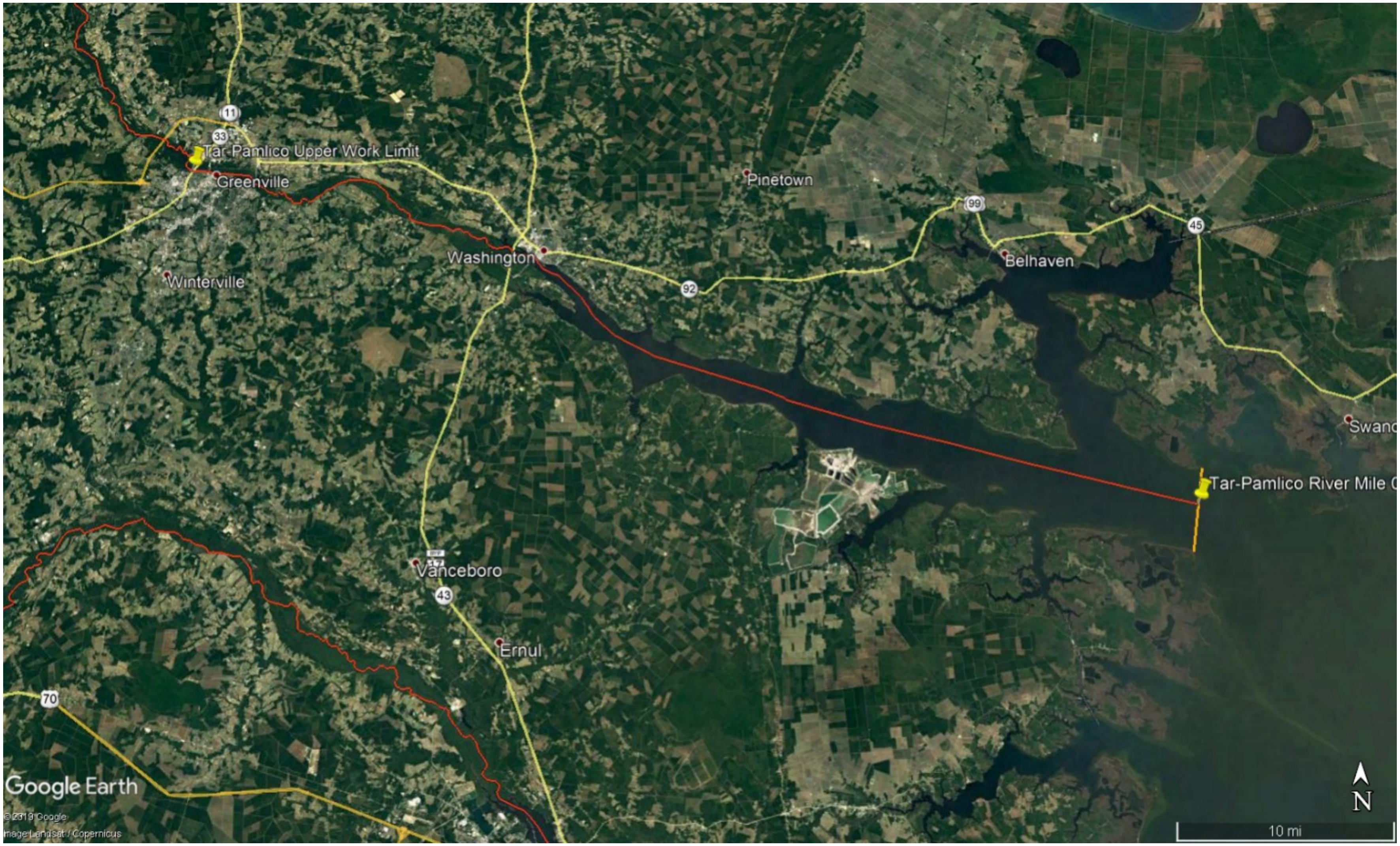


Figure 57. Tar-Pamlico River
River mile 0 is marked with a yellow pin and yellow line. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

Figure 58. Neuse River
River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel in the Neuse River. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

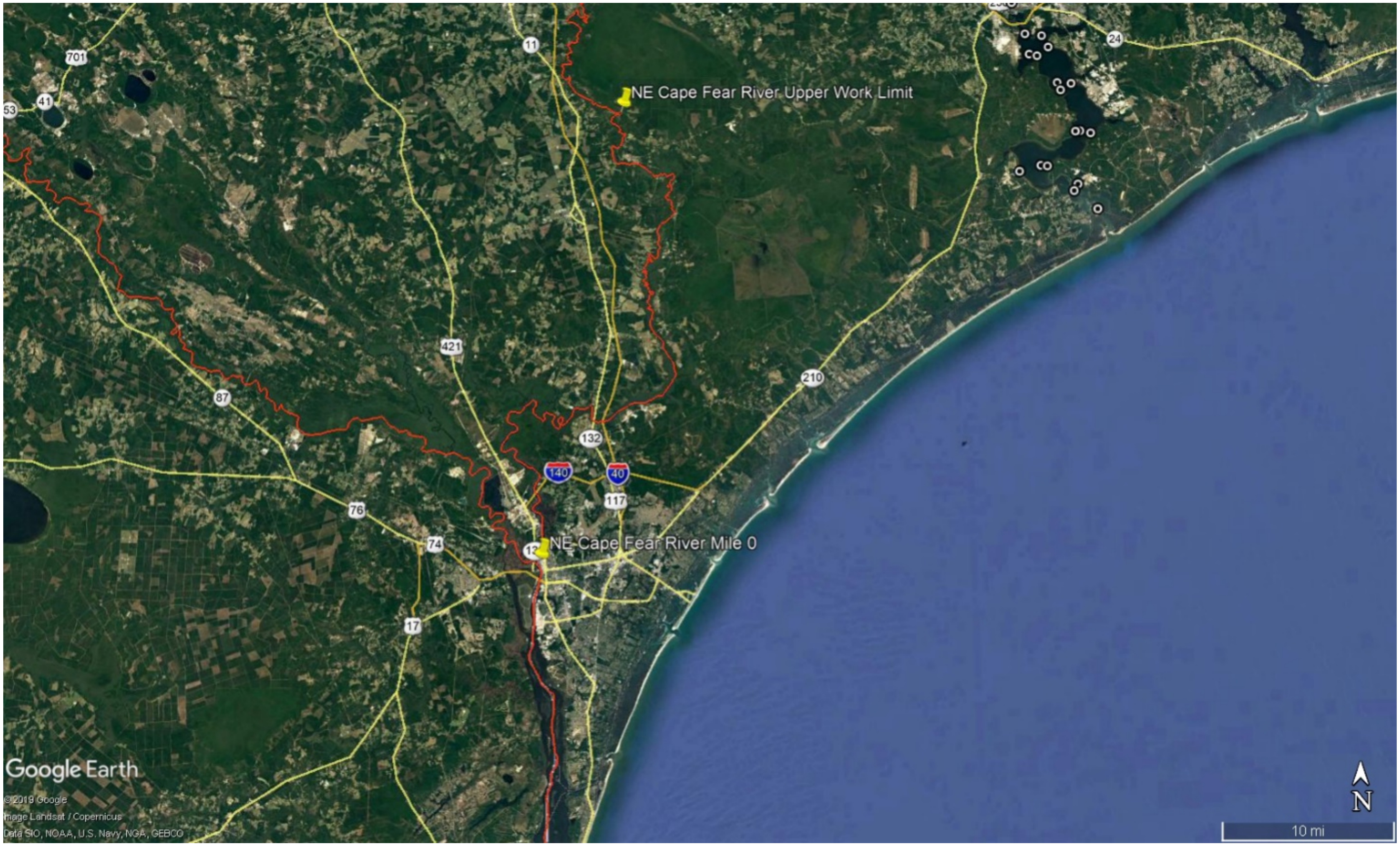


Figure 59. Northeast Cape Fear River
River mile 0 is marked with a yellow pin and yellow line. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

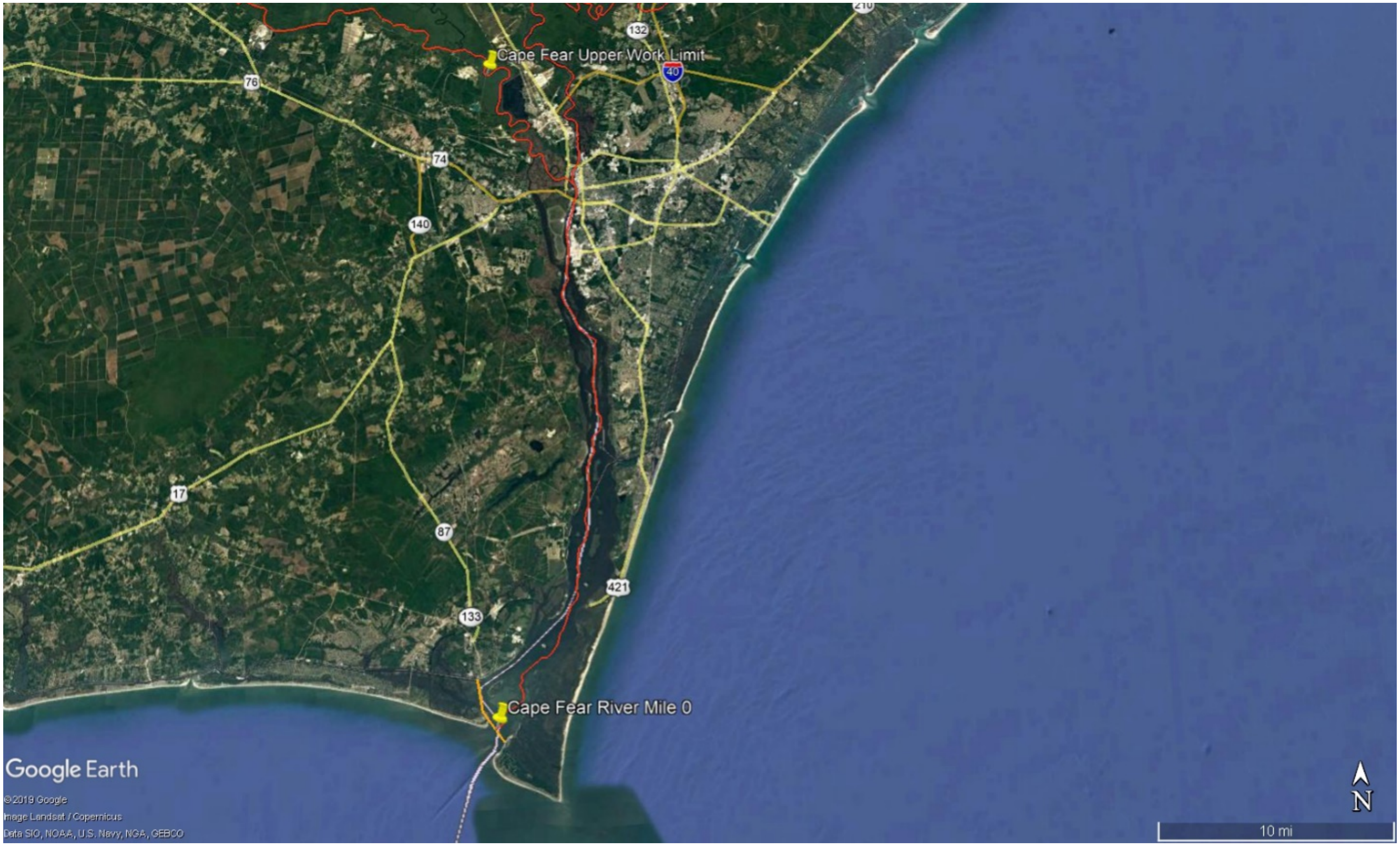


Figure 60. Cape Fear River
River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel in and around Wilmington Harbor and the Cape Fear River. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

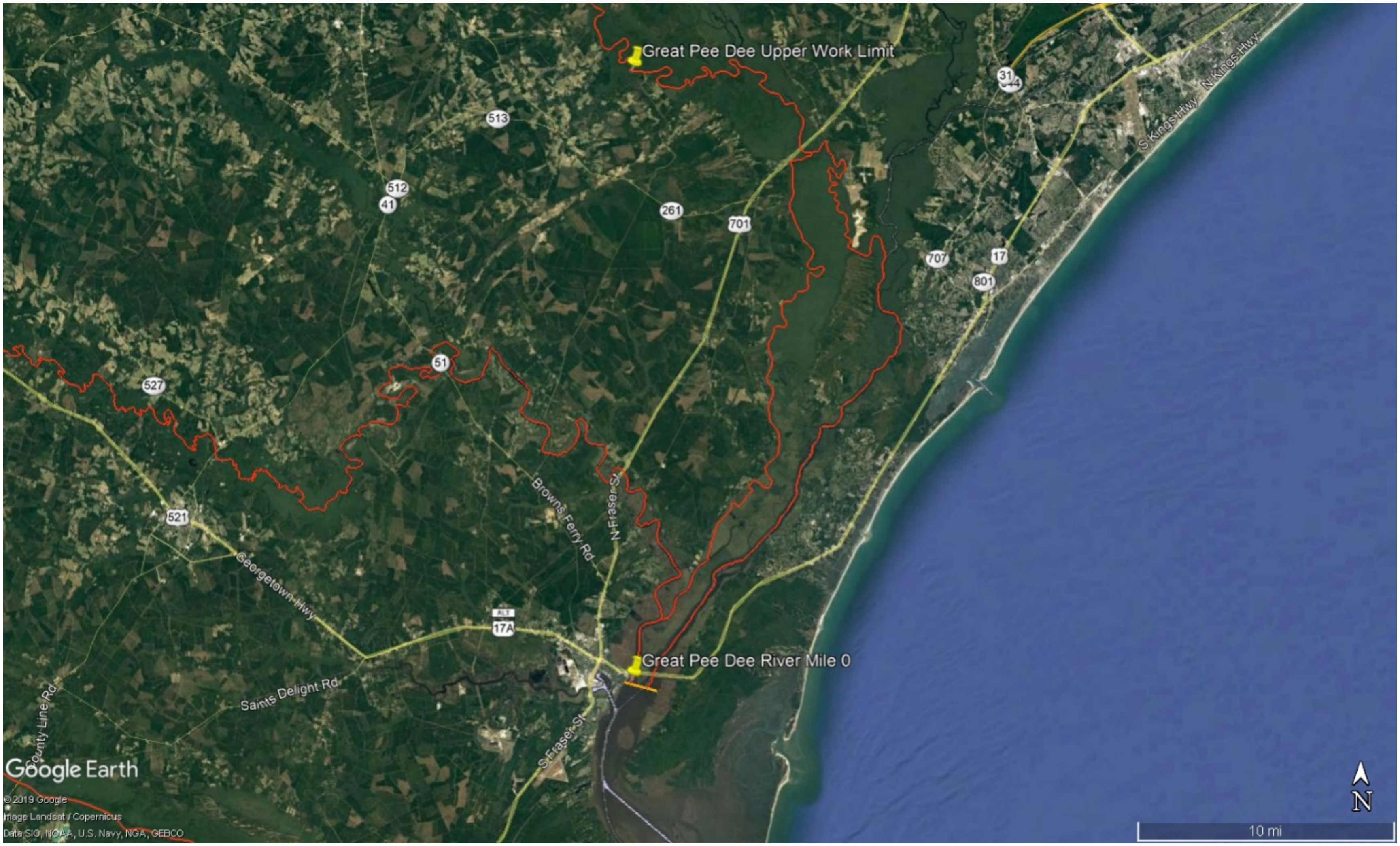


Figure 61. Great Pee Dee River
River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel in and around Winyah Bay. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

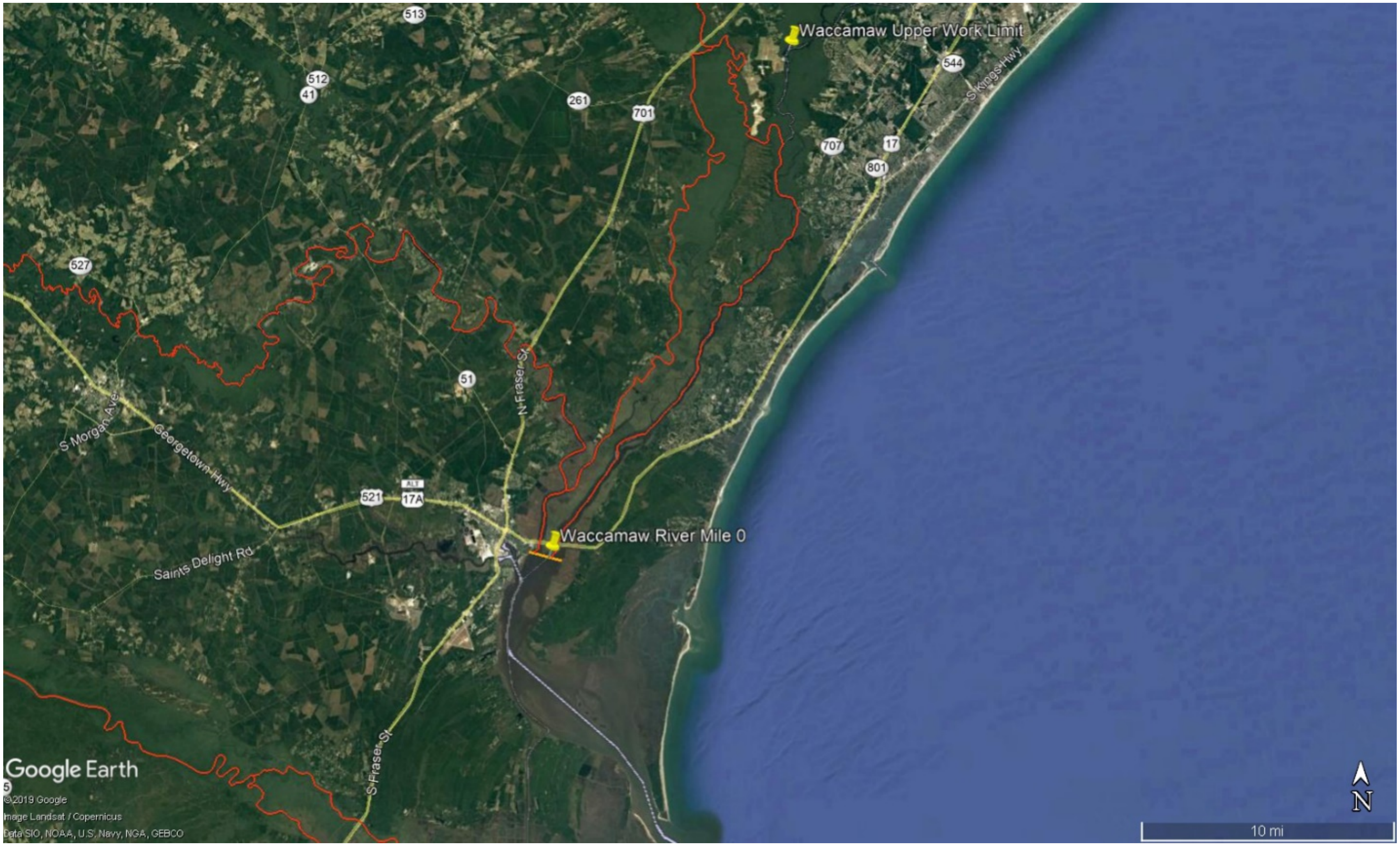


Figure 62. Waccamaw River

River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel in and around Winyah Bay. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

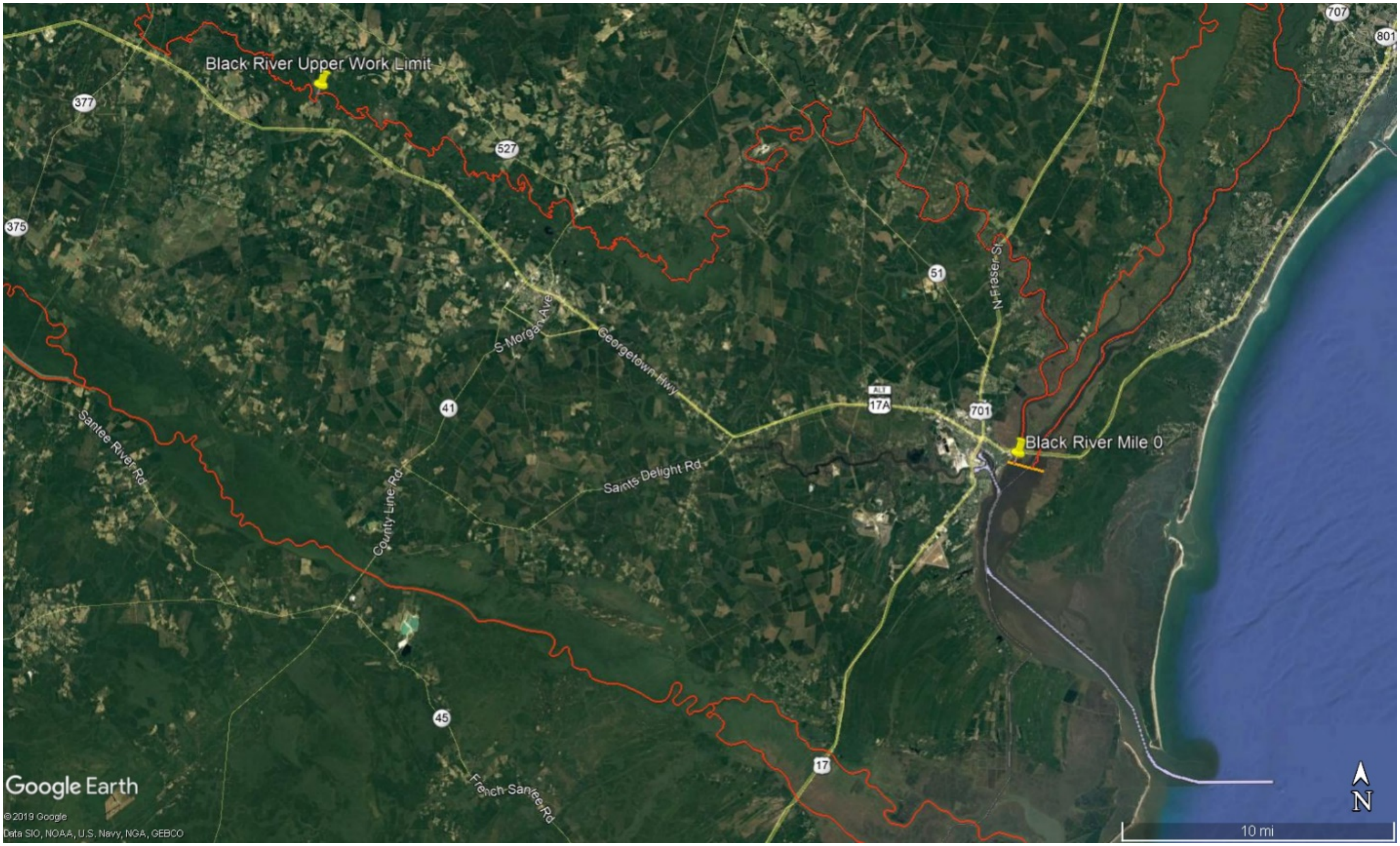


Figure 63. Black River
River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel in and around Winyah Bay. The upper river limit is also marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

Figure 64. North/South and Mainstem Santee River with the Rediversion Canal

River mile 0 for the South Santee and North Santee rivers are marked with a yellow pin and yellow line. Yellow pins also mark the confluence of the South Santee and North Santee river with the mainstem of the Santee River. The upper work limit on the Santee River is also marked, along with the St. Stephen Powerhouse, and the confluence of the Rediversion Canal with the mainstem of the Santee River. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

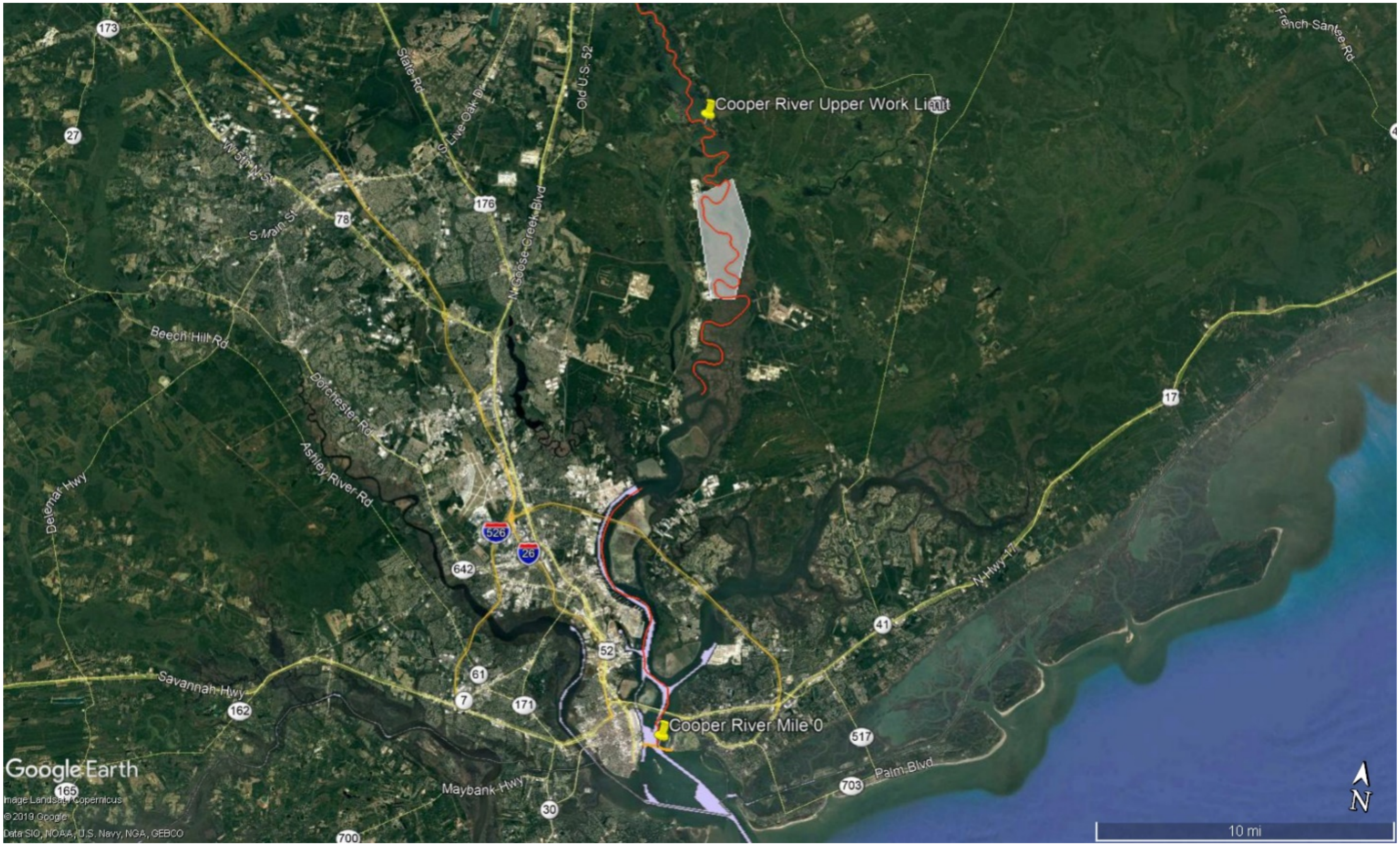


Figure 65. Cooper River

River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel in and around Charleston Harbor. The white-shaded box is the aggregation area. The upper river limit is marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

Figure 66. North/South and Mainstem Edisto River
River mile 0 for both the South Edisto and North Edisto rivers is marked with a yellow pin and yellow line. The upper work limit for the Edisto River marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

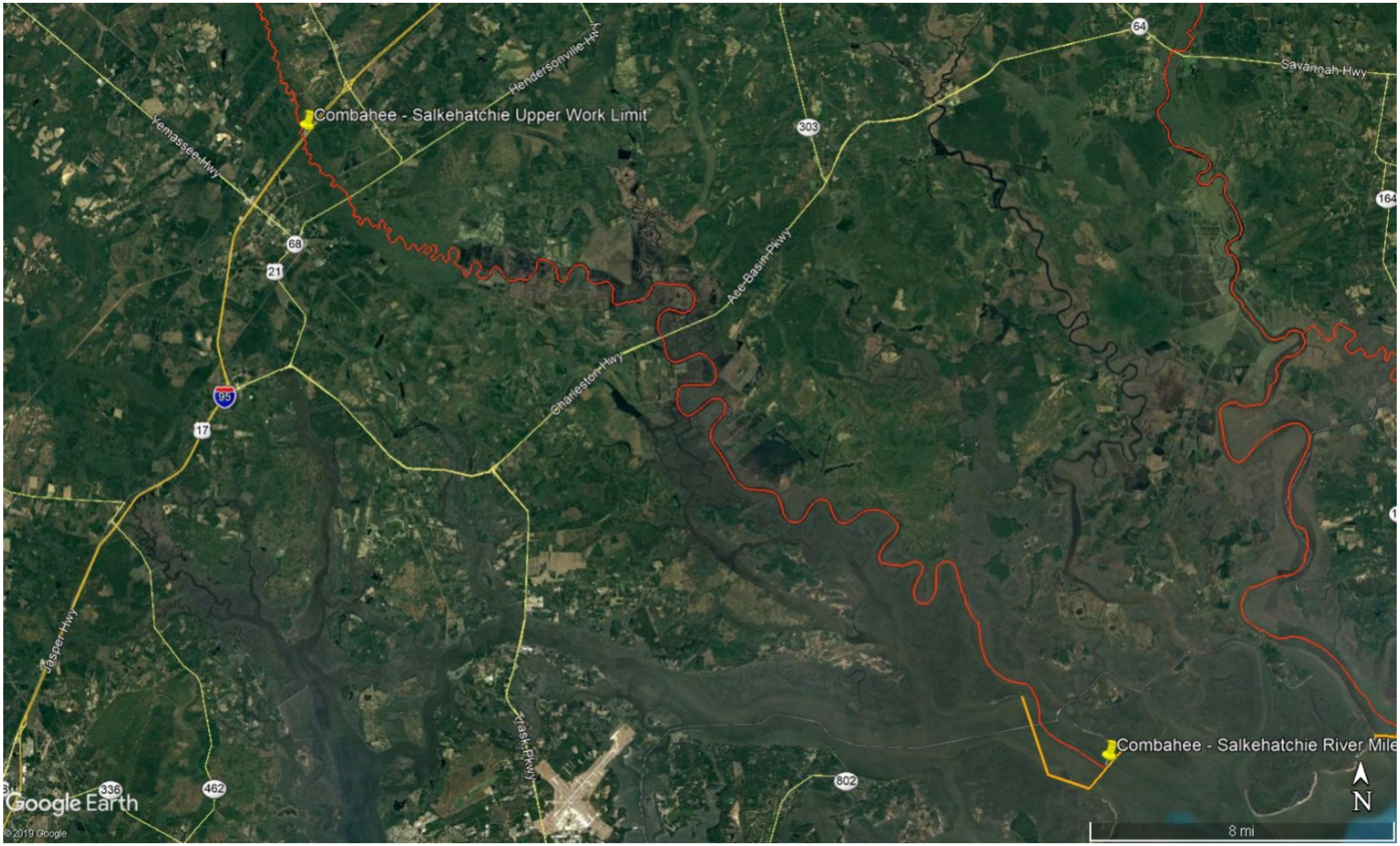


Figure 67. Combahee-Salkehatchie River
River mile 0 is marked with a yellow pin and yellow line. The upper work limit is marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

Figure 68. Savannah River
The blue line is the USACE maintained channel and the shaded box is the aggregation area. River mile 0, the upper river limit, and Atlantic sturgeon critical habitat are not shown.

Figure 69. Ogeechee River

River mile 0 is marked with a yellow pin and yellow line. The upper work limit is marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line. The shaded box is the aggregation area.

Figure 70. Altamaha River Sturgeon Seasonal Aggregation Area
River mile 0 is marked with a yellow pin and yellow line. The IWW is marked with a yellow pin and blue line. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line. The shaded box is the aggregation area.

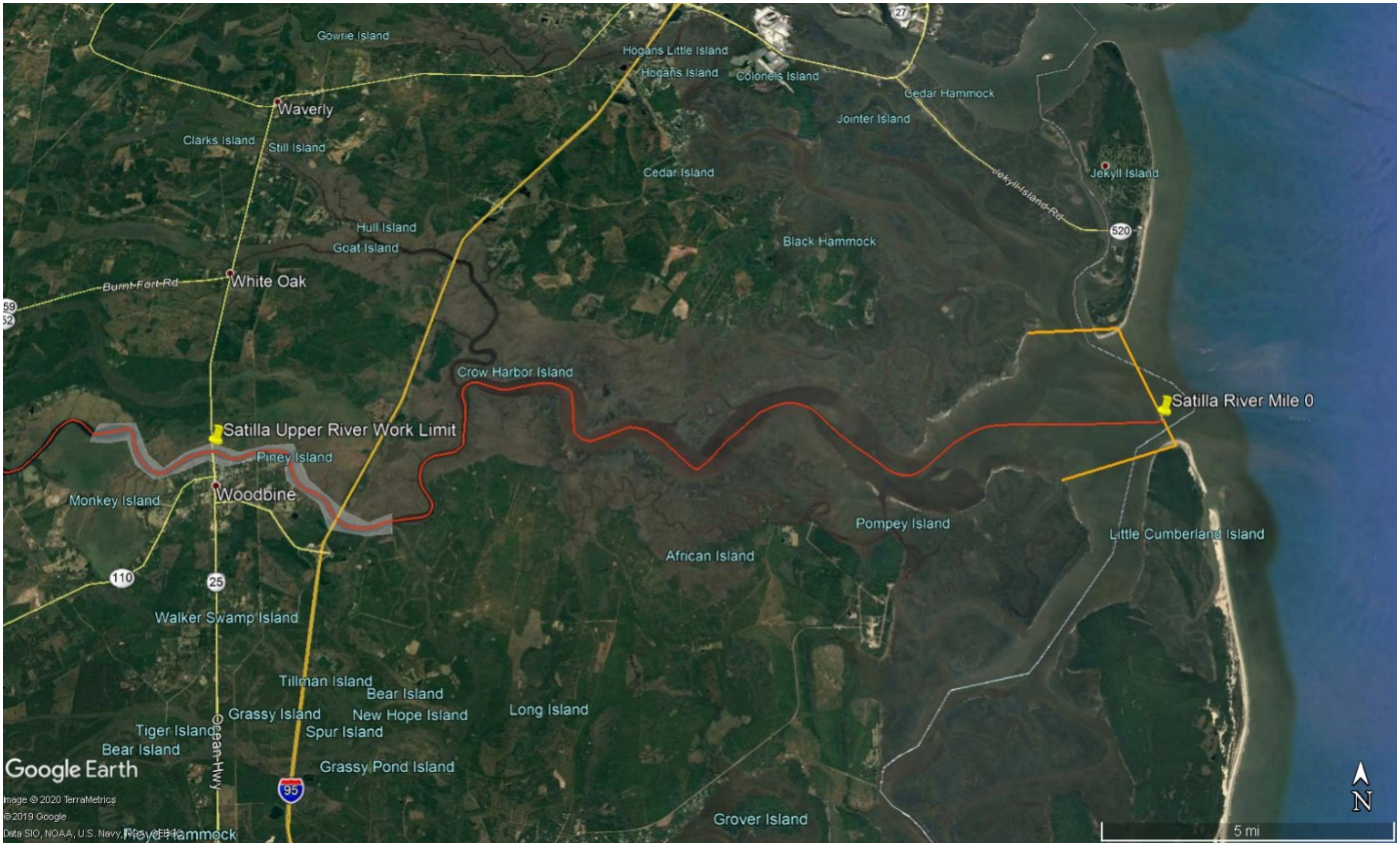


Figure 71. Satilla River
River mile 0 is marked with a yellow pin and yellow line. The upper work limit is marked with a yellow pin. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line. The white-shaded box is the aggregation area.

Figure 72. St Marys River

River mile 0 is marked with a yellow pin and yellow line. The purple-shading indicates the USACE maintained channel to and including the IWW to Kings Bay Base. The white-shaded box is the aggregation area. The upper river limit is marked with a yellow pin at I-95. River stretches designated as Atlantic sturgeon critical habitat are denoted as a red line.

4 Cutterhead Dredging Monitoring in Sturgeon Rivers

Cutterhead dredging in sturgeon rivers will be monitored for take of sturgeon in the sections of rivers and during the times identified in Table 56 with the dredging restriction B or C as described in Sturgeon PDC Section 3.

- Monitoring: An approved PSO (described in the PSO PDCs in Appendix H) or Government Quality Assurance personnel or contractor can be appointed as a designated observer in charge of monitoring the upland discharge/disposal area used for cutterhead dredging in sturgeon rivers.
- Training: The designated observer will have training in identification of fish parts. Information on sturgeon identification is provided PSO PDCs in Appendix H. The designated observer will also train all crew of the presence of sturgeon and that all sturgeon or suspected fish parts will be reported to the designated observer.
- Timing: The designated observer will check for fish or parts at the overflow end of the disposal pond and any other areas where debris collects at least twice per day.
- Collection of fish by a designated crew member: The designated crew member will collect, document, and photograph all parts identified as or suspected to be fish or fish parts according to the handling guidance in the PSO PDCs in Appendix H. Collected fish parts must then be passed to an approved PSO or a trained biologist for identification.
- Collection of fish by an approved PSO/trained biologist: The PSO/trained biologist will collect, document, photograph, and report all parts identified as sturgeon or suspected to be sturgeon parts according to the handling guidance in the PSO PDCs in Appendix H.
- Genetic Sampling: All Atlantic sturgeon or suspected sturgeon parts will have a genetic sample taken and submitted for testing to identify if the fish collected was a sturgeon and the Atlantic sturgeon DPS according to the guidance in the PSO PDCs in Appendix H.
- Reporting: All sturgeon collected will be reported according to the guidance in the PSO PDCs in Appendix H and the 2019 SARBO Section 2.9 of the 2020 SARBO.

Appendix F. 2020 SARBO USACE and BOEM North Atlantic Right Whale Conservation Plan

The Conservation Plan for the North Atlantic right whale includes "Avoidance Measures", which act as PDCs that apply to all projects that occur where and when North Atlantic right whales may be present (as defined in the Plan). The Plan is hereby incorporated into the 2020 SARBO and is included in this Appendix. The avoidance measurements are in addition to any other applicable PDCs outlined in Appendix B-H and the requirements outlined in the 2020 SARBO.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but have substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the Opinion, will require individual Section 7 consultation. Any area previously authorized or permitted to be dredged or have material placed within the action area and analyzed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

1 Introduction

1.1 Background

Section 7(a)(1) of the ESA requires all federal agencies to use their authority as appropriate to carry out programs for the conservation (i.e., recovery) of threatened and endangered species. For about 3 decades, the USACE has worked with NMFS and state conservation agencies to identify and resolve endangered species and ecosystem management concerns resulting from dredging in the Southeastern United States. The endangered North Atlantic right whale has been a particular focus of conservation efforts in the Southeastern United States, where the species has its only known calving habitat. The North Atlantic right whale Conservation Plan outlines management measures that the USACE SAD and BOEM (where applicable) will implement within its area of responsibility to create an ESA Section 7(a)(1) conservation program for North Atlantic right whales. The USACE SAD developed this program through consultation with the NMFS Southeast Regional Office. The North Atlantic Right Whale Conservation Plan complies with USACE Environmental Operating Principles, Civil Works Ecosystem Restoration Policy (ER 1165-2-501), and supports the conservation intent of the Marine Mammal Protection Act (16 U.S. Code Chapter 31).

1.2 Purpose and Scope

This Conservation Plan (Plan) is prepared pursuant to Section 7(a)(1) of the Endangered Species Act, as amended, which requires all federal agencies to use their authority to carry out programs for the conservation (i.e., recovery) of endangered and threatened species. The purpose of the Plan is to describe USACE and BOEM (where applicable) management measures to conserve the North Atlantic right whale within the action area. The geographic area (action area) covered by this plan includes the coastal waters of the Atlantic Ocean from the Virginia/North Carolina border south to Cape Canaveral, Florida – referred to in the rest of this document as “the range of North Atlantic right whale”.

This plan includes systems to detect the presence of whales, alert vessels operating in the area, and avoidance and minimization measures for projects covered under the 2020 SARBO that reduce the risk of a vessel strike if a whale is detected in the area. Implementation of these management measures will minimize the risk of North Atlantic right whale/dredge vessel interactions and will contribute to North Atlantic right whale recovery. Funding for management measures will be provided by USACE and is contingent upon annual appropriations. These measures will be executed in order for USACE and BOEM projects to be covered under the 2020 SARBO, especially if the project occurs within the range of North Atlantic right whale during the timeframe they are expected to present, as outlined in this Plan. If USACE funding is not available to execute one or more of the conservation measures, then USACE and BOEM will confer with NMFS on the appropriate next steps.

2 Conservation Measures

2.1 Established Right Whale Early Warning System (EWS)

The USACE has entered into an agreement with the U.S. Navy, U.S. Coast Guard, and NMFS over 2 decades ago that sets forth a framework for implementing North Atlantic right whale EWS aerial surveys. BOEM is not party to this agreement but will leverage the data accordingly. The EWS surveys are flown on days with suitable weather (good visibility and sea state of not more than Beaufort sea state 3)[90], from December 1 through March 31. East-west transects spaced 3 nautical miles apart are flown to visually detect North Atlantic right whale in the vicinity of the commercial shipping ports of Brunswick, Georgia, and Fernandina Beach and Jacksonville, Florida (Figure 73). Submarine Base Kings Bay and Naval Station Mayport are also located within the survey area. Transect locations may be altered, as necessary, to maximize North Atlantic right whale detection. The EWS has likely decreased the number of vessel and whale interactions for not only federal vessels, but also commercial vessels. The use of current technology to notify vessels in the area of whale presence enables vessel operators to either avoid the area, increase their vigilance, or decrease their speed based on the known presence of right whales. While the number of vessels associated with USACE dredging is not significant, USACE is willing to minimize the amount of activity in an area either when whales are known to be present or in areas when whales are likely to be present in order to decrease the probability of interactions between whales and vessels. Adding to the information about North Atlantic right whale – through aerial surveys (or other appropriate survey techniques) – adds to the overall knowledge about their behavior and their temporal and spatial distribution. Marine species location information will be contributed to established databases (e.g. North Atlantic Right Whale Consortium, Ocean Biogeographical Information System, etc.) and thereby be made available to managers and researchers (e.g. for inclusion in modeling efforts like (Gowan and Ortega-Ortiz 2014; Roberts et al. 2016). Relevant new/ongoing BOEM North Atlantic right whale research will also be made available as appropriate.

[90] Details of Beaufort Wind Scale can be found at https://www.spc.noaa.gov/faq/tornado/beaufort.html

During EWS surveys and other relevant BOEM research activities (where applicable), locations of detected North Atlantic right whales are relayed in near real time to subscribers of geographic-specific listservs to which Whale Alerts are issued. Prior to each dredge project, funding agency personnel and commercial and marine users may subscribe to geographic-specific listservs to receive Whale Alerts. Whale Alerts are distributed via text messages and emails to subscribers. Other means for distributing sighting information includes VHF radio, minimum storage regeneration return messages, NAVigational TEleX, and internet websites, etc. In addition to ship collision avoidance and population monitoring, these surveys collect important data on population status, calf production, distribution, and habitat use.

Aerial surveys and other relevant BOEM research will also include reported sightings of all observed notable species including other ESA-listed species observed (e.g., giant manta ray and leatherback sea turtles), as long as it does not detract from the main purpose of reporting North Atlantic right whale. Reporting on other species allows for a better understanding of their temporal and spatial distribution as well.

Detailed EWS reports and North Atlantic right whale sighting information can be found at:
http://sero.nmfs.noaa.gov/protected_resources/right_whale/seus_sightings/index.html

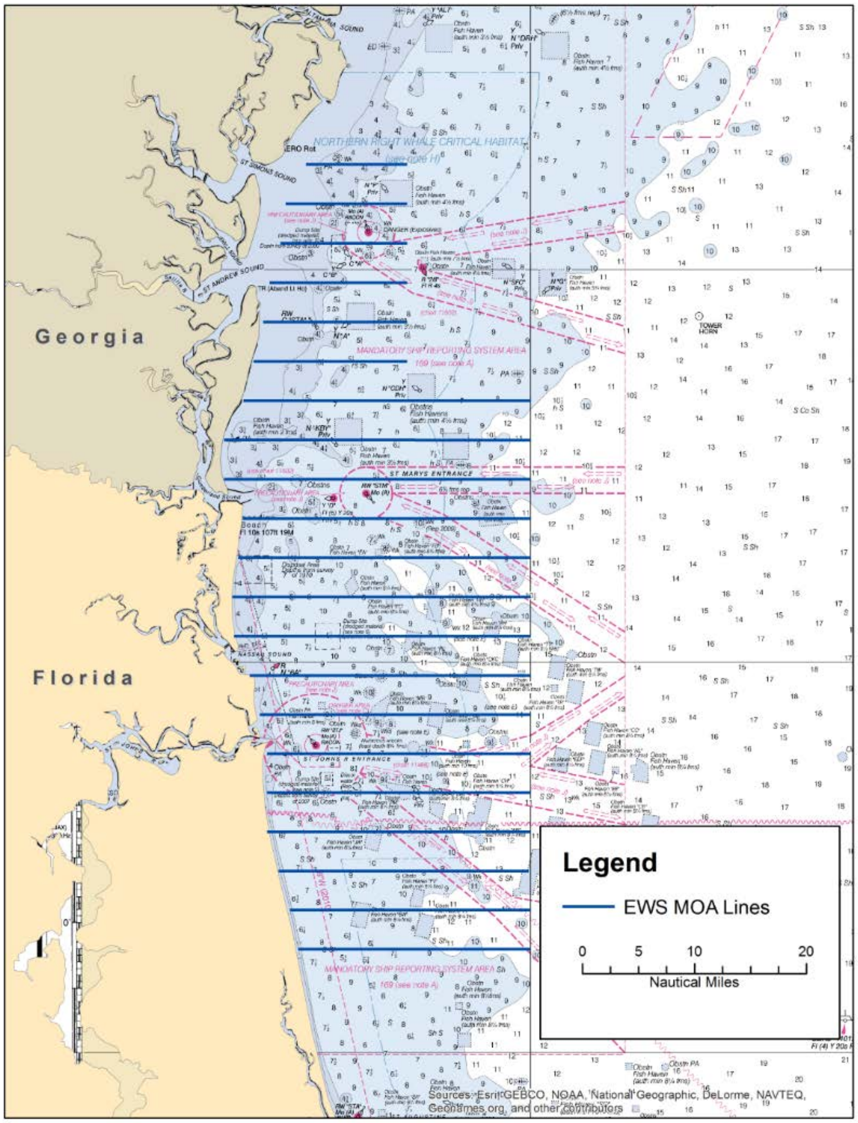


Figure 73. An Example of the Early Warning System Tracklines (blue lines).

2.2 Expanded Data Collection on Temporal and Spatial Distribution of North Atlantic right whale

The USACE will fund additional aerial survey to learn more about the temporal and spatial distribution of North Atlantic right whales off the Southeast Atlantic coast. This additional investment will expand the footprint of aerial survey coverage Brunswick, Georgia through North Carolina from 15 November through 15 April, annually. This additional aerial support will be on par with 860 aerial survey hours; however, detection methods may change contingent upon new information and consultation with species experts, including NMFS. This will supplement work that is already being accomplished. As in the North Atlantic Right Whale Conservation Plan, locations of detected North Atlantic right whales are relayed in near real time to subscribers of geographic-specific listservs to which Whale Alerts are issued (as described above). In addition to ship collision avoidance and population monitoring, these surveys will collect important data about migration, distribution, and habitat use, and contribute to long term databases. They will also provide information about whale presence to aid in implementation of the avoidance measures below. Aerial surveys will also report observed notable species including other ESA-listed species (e.g., giant manta rays), as long as it does not detract from the main purpose of detecting and reporting North Atlantic right whale.

2.3 Avoidance Measures

The USACE and BOEM (as appropriate) will implement the North Atlantic right whale Conservation Plan within an Atlantic coastal action area extending from the Virginia/North Carolina border south to Cape Canaveral, Florida, during the North Atlantic right whale migration and calving season from November 1 through April 30. The following measures are in addition to the other 2020 SARBO PDCs and the Programmatic Implementation, Tracking, and Reporting requirements outlined in the 2020 SARBO Section 2.9. Avoidance measures consist of the following:

NARW.1 Dredge Project Scheduling

The Risk-Based Adaptive Management Process (outlined in the 2020 SARBO Section 2.9.2.2) explains the process used to determine where, when, and how projects will be completed to minimize the risk to ESA-listed species.

- Hopper dredging and projects requiring survey vessels over 33-ft in length will be scheduled, to the maximum extent practicable, outside of North Atlantic right whale migration and calving season to avoid impacts to North Atlantic right whales, including reproducing females and newborn calves. Other information that will be considered includes where material is to be placed and whether the timing of the placement would be high risk for other listed species (e.g. sea turtles).
- Prior to each fiscal year (which is also prior to North Atlantic right whale calving season), the USACE and BOEM will notify NMFS of anticipated dredge projects planned in the South Atlantic and for those taking place in the Southeast Seasonal Management Area (SMA), shown in Figure 74, which minimization and avoidance measures will be accomplished. This information will be provided as part of the Risk-Based Adaptive Management Process (outlined in the 2020 SARBO Section 2.9.2.2).

NARW.2 Captains and crew of USACE and USACE vessels, contracted vessels, and PSO requirements:

- All on-site project personnel associated with a project covered under SARBO including the vessel captain, crew, and PSO on dredge vessels, survey vessels, and all supporting vessels over 33-ft in length will be instructed on the presence of North Atlantic right whale and other ESA-listed species and the requirements to observe, avoid, and report North Atlantic right whale in the area. The required distances that vessels must maintain from ESA-listed species, PSO observer coverage requirement for 100% monitoring on hopper dredging and relocation trawling, and reporting requirements are defined in the 2020 SARBO (PSO PDCs in Appendix H).

- All captains of dredges, relocation trawlers, survey vessels, and support vessels over 33-ft in length will provide a text message address (that is capable of receiving short emails as text messages) to receive real-time whale alerts throughout the calving season. The text message address will be provided to nmfs.ser.rw.subscribe@noaa.gov at least 14 days prior to the start of dredging or annually on November 1 if the vessel is utilized year round.
- The dredging company contractor for each project, before the start of dredging, will provide a single whale observer email address to receive aerial survey-related notifications (status, fly/no-fly plans, etc.) that will be immediately sent to all active vessels working in water for the project. The email address will be provided to nmfs.ser.rw.subscribe@noaa.gov and be confirmed annually, prior to each North Atlantic right whale calving season.
- All hopper dredges and relocation trawlers will have onboard dedicated daytime[91] PSOs that meet the qualifications provided by NMFS and detailed in SARBO with at-sea, large whale identification experience to conduct observations for the presence of whales and all other ESA-listed species. The PSO will have the primary duty of observation when the vessel is underway. Observers will be onboard dredges and will alternate to reduce observer fatigue. As needed, a crew member on the bridge will assist the PSO with whale observation duty while the vessel is underway. The PSO will provide crew members with appropriate training for large whale observation. Hopper dredges will submit an endangered species watch plan detailing how the requirements to minimize the risk of a North Atlantic right whale/dredge vessel interaction will be accomplished. The watch plan may be a component of an environmental protection plan.
- The PSO will note all sightings of ESA-listed species and marine mammals according to the reporting requirements in the 2020 SARBO Section 2.9and by submitting all necessary forms and information to the ODESS. All ESA-listed marine mammals spotted will also be immediately reported by calling 1-877-WHALE_HELP.
- All project vessels will carry operational automatic identification system transmitters as required by the U.S. Coast Guard. Transmitters will be powered on and transmitting while vessels are underway and NMFS will be provided the vessel name and vessel tracking number (maritime mobile service identities) so that all vessels operating under SARBO can be tracked and confirm compliance with this Plan. Vessel tracking numbers will be recorded in ODESS and emailed to NMFS at **SERO.Dredge@noaa.gov** for all vessels over 33-ft in length operating from the Virginia/North Carolina border south to Cape Canaveral, Florida, during the North Atlantic right whale migration and calving season from November 1 through April 30 (see SARBO General PDCs, Section 2, Appendix A).

[91] Daytime is defined as 30 minutes before sunrise to 30 minutes after sunset.

NARW.3 Vessel Speed Requirements

Speed requirements must be followed if a North Atlantic right whale has been spotted or reported in the area as defined below. North Atlantic right whale presence may be determined by observers on the vessel, reports from aerial surveys, EWS, or confirmed public sighting reports. All captains are required to use daily available information and reports on the presence of North Atlantic right whales and aerial survey activities in the project area. These speed restrictions apply to all vessels associated with a project covered under SARBO.

- Vessels over 65 ft in length: When a whale is observed or reported within 38 nmi of dredge or support vessels, vessels must slow to 10 knots or slowest safe navigable speed for 36 hours[92] or until next North Atlantic right whale survey when no whales are observed, whichever is shorter.
- Vessels 33- 65 ft: When a whale is observed or reported within 38 nmi of dredge or support vessels which are within the Southeast SMA, vessels must slow to 10 knots or slowest safe navigable speed until the next North Atlantic right whale survey when no whales are observed. Vessels may resume speed once outside the Southeast SMA. Outside of the Southeast SMA, but within the range of North Atlantic right whale habitat, the vessel must slow to 10 knots for 36 hours, or until the next North Atlantic right whale survey when no whales are observed.

Table 58. Vessel Speed Requirements, if a North Atlantic right whale is spotted or reported[93] within 38 nmi of the vessel:

Vessel Size	Within the range of North Atlantic right whale[94]	Within the SMA (Figure 74)
33-65 ft	• All vessels must slow to 10 knots or slowest safe navigable speed • Slow speed maintained for 36 hours[95] or until next North Atlantic right whale survey when no whales are observed, whichever is shorter.	• All vessels must slow to 10 knots or slowest safe navigable speed • Slow speed maintained until the next North Atlantic right whale survey when no whales are observed (no time limit). • Vessels may resume speed once outside the Southeast SMA.
Over 65 ft	• All vessels must slow to 10 knots or slowest safe navigable speed • Slow speed maintained for 36 hours or until next North Atlantic right whale survey when no whales are observed, whichever is shorter.	

[92] Contractors will be required to use daily available information on the presence of North Atlantic right whales and aerial survey activities in the project area.

[93] Contractors will be required to use daily available information on the presence of North Atlantic right whales and aerial survey activities in the project area.

[94] Currently defined as occurring from the Virginia/North Carolina border south to Cape Canaveral, Florida

[95] Contractors will be required to use daily available information on the presence of North Atlantic right whales and aerial survey activities in the project area.

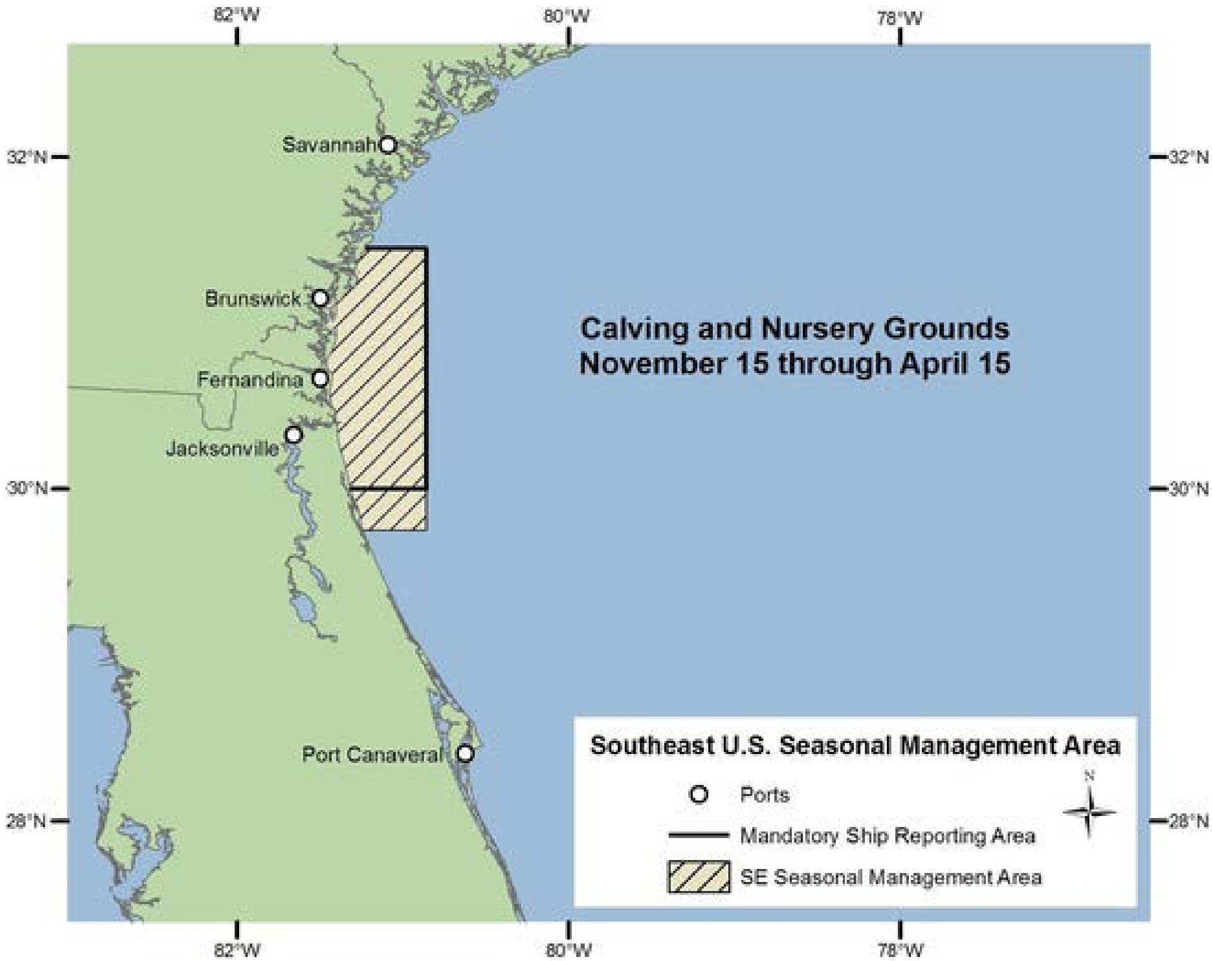


Figure 74. Seasonal Management Area for North Atlantic Right Whale[96]

2.4 Volunteer Sighting Network

In addition to the contributions that USACE makes toward the EWS aerial surveys, the USACE Jacksonville District also provides annual funding for the Marineland Right Whale Project volunteer sighting network. The Marineland Right Whale Project uses shore-based volunteer spotters to look for right whales between St. Augustine and Ponce Inlet, Florida. The Marineland Right Whale Project not only relays additional whale sightings to the EWS system and the New England Aquarium's North Atlantic right whale sighting database, but project staff also provide North Atlantic right whale information to the public.

[96] This image includes the mandatory ship reporting area, which is defined under the Mandatory Ship Reporting Systems Rule (69 FR 58066). This rule states that when ships greater than 300 gross tons enter 2 key right whale habitats (located off the northeast U.S. and off the southeast U.S.), they are required to report to a shore-based station. In return, ships receive a message about right whales, their vulnerability to ship strikes, precautionary measures the ship can take to avoid hitting a whale, and locations of recent sightings. These requirements are in addition to and separate from the requirements of the 2019 SARBO.

Appendix G. Geophysical and geotechnical surveys PDCs

The PDCs in Appendix G apply to all projects that use G&G survey equipment. These requirements are in addition to any other applicable PDCs outlined in Appendix B-Appendix H and the requirements outlined in the 2020 SARBO.

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but have substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the 2020 SARBO, will require individual Section 7 consultation. Any area previously authorized or permitted to be dredged or have material placed within the action area and analyzed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

1 2020 SARBO Geophysical and Geotechnical (G&G) PDCs

G&G PDCs that apply to all geotechnical surveys

GG.1 For geotechnical surveys, the vibrahead will not be operated until the vibracore platform makes contact with the seabed and core barrel makes contact with the seafloor. The vibrahead will not be operated when vibracore platform is being retrieved.

G&G PDCs that apply to all geophysical surveys

GG.2 No geophysical surveys will occur at night or during periods of low visibility.

GG.3 The minimum number of geophysical sources possible will be used to obtain the necessary geophysical data and the acoustic source will be deactivated when not acquiring data or preparing to acquire data, except as necessary for testing.

GG.4 Only electromechanical sources will be used during geophysical surveys. Electromechanical sources will be limited to boomers, chirp sub-bottom profilers, side-scan sonars, and single beam, interferometric, or multibeam depth sounders.

- Survey equipment will be operated at the lowest power setting, narrowest beamwidth, and highest frequency possible to fulfill data needs and to effectively reduce exposure and received sound levels.
- Boomers and chirp sub-bottom profilers must be operated below 205 dB re 1 μPa (rms).

- Single beam depth sounders will be operated no lower than 24 kHz.[97]
- Side-scan sonars, interferometric, and multibeam depth sounders will be operated above 160 kHz
- No airguns or other deep-penetrating geophysical instruments are allowed under 2020 SARBO.

Geophysical Survey Requirements for Sturgeon

GG.5 Sturgeon rivers: Geophysical surveys will not occur upstream of the upper river limits of sturgeon rivers as defined in Section 3 in Appendix E.

Geophysical survey requirements for ESA-listed whales

GG.6 Between November 1st and April 30th, no geophysical surveys will be conducted that operate between 10 Hz-22 kHz where North Atlantic right whales may be present from the North Carolina/Virginia border south to Cape Canaveral, Florida.

GG.7 Geophysical surveys will cease operations if an ESA-listed whale is observed within 500 yards (460 m) of the active geophysical survey equipment. Subsequent restart of the equipment will only occur once all ESA-listed whales are more than 500 yards away from – or have not been observed or detected within 500 yards of – the equipment for a minimum of 30 minutes. If an ESA-listed whale is observed, a detailed observation will be provided to NMFS within 24 hours including:

a. All species observed, highlighting ESA-listed species
b. Activity of vessel when sighting occurred (e.g., survey vessel movement details such as speed and direction and survey equipment in use at the time)
c. State if work ceased when a species was observed.
d. Species and certainty of identification (i.e., positive identification, most likely, best guess);
e. Photograph of the animal for identification, if possible
f. total number of animals;
g. number of calves and juveniles (if applicable/distinguishable);
h. description (as many distinguishing features as possible) of each individual seen, including length, shape, color and pattern, scars or marks, shape and size of dorsal fin, shape of head, and blow characteristics
i. direction of animal's travel relative to the vessel (drawing preferable); and
j. behavior (as explicit and detailed as possible; note any observed changes in behavior).

[97] PDC revised in July, 30 2020, to clarify the use of single beam sonar. See Appenix L, Revision History for more information.

Appendix H. Handling and Reporting Protocol for ESA-listed Species Observed or Encountered and Protected Species Observer (PSO) Roles and Responsibilities

All ESA-listed species that are observed or encountered during any activity covered under the 2020 SARBO, will be handled and reported as described in this Appendix, referred to as the PSO PDCs. These PDCs outline the requirements of vessel crew to report observations and for the PSO to observe for and handle ESA-listed species captured during dredging or relocation trawling. These requirements are in addition to any other applicable PDCs outlined in Appendix B-Appendix G and the requirements outlined in the 2020 SARBO. Vessel crew and PSOs working on projects covered under the 2020 SARBO should also be aware of the conditions in the PDCs that are applicable to the project upon which they are working on under the 2020 SARBO. Modifications to the handling procedures may be necessary to improve safe handling practices for both crew and animals. The current handling guidance (PSO PDCs) is available at (**SERO.Dredge@noaa.gov**).

Alternative review: In limited instances, a project may be authorized under the 2020 SARBO if it does not adhere to all applicable PDCs, under the Alternative Process for Project Specific Review and Inclusion of Projects with Substantially Similar Effects outlined in Section 2.9.5 of the 2020 SARBO. As described in the 2020 SARBO, projects that do not strictly comply with all applicable PDCs, but have substantially similar effects, may be authorized under 2020 SARBO if the project undergoes separate review and approval by NMFS prior to beginning work. Projects that cannot meet all relevant PDC requirements or that do not fit under the alternative review process outlined in Section 2.9.5 of the Opinion, will require individual Section 7 consultation. Any area previously authorized or permitted to be dredged or have material placed within the action area and analyzed in a separate individual Section 7 consultation may be maintained to the same dredge or fill template under this Opinion if it meets all of the PDCs of this Opinion.

1 Observations and Reporting Observations of ESA-listed Species

This outlines how staff operating on a project covered under the 2020 SARBO will respond to ESA-listed species that are observed, but with no physical interaction occurring with the animal.

OBSERVE.1 For generally stationary construction with work contained to a specific project area, such as mechanical dredging equipment:

- All personnel working on the project will report ESA-listed species observed in the area to the on-site crew member in charge of operations.
- Operations of moving equipment will cease if an ESA-listed species is observed within 150 ft of operations by any personnel working on a project covered under this Opinion (e.g., sea turtles, sturgeon, elasmobranchs [smalltooth sawfish, giant manta ray, scalloped hammerhead shark, oceanic white tip shark] or ESA-listed marine mammal).
- Activities will not resume until the ESA-listed species has departed the project area of its own volition (e.g., species was observed departing or 20 minutes have passed since the animal was last seen in the area).

OBSERVE.2 For a vessel underway, such as a hopper dredge or support vessel, traveling within or between operations must follow speed and distance requirements, defined below, while ensuring vessel safety:

- All personnel working onboard will report ESA-listed species observed in the area to the vessel captain.
- If an ESA-listed species is spotted within the vessel's path, initiate evasive maneuvers to avoid collision.
- If a North Atlantic right whale is spotted, slow to 10 knots and maintain a distance of at least 1,500 ft in accordance with the North Atlantic Right Whale Protection Rule (62 FR 6729 provides a distance of 500 yards, which is equal to 1,500 ft) and report the observation to 1-877-WHALE-HELP. Resumption of speed should be according to the North Atlantic Right Whale Conservation Plan (Appendix F).
- If a whale (other than a North Atlantic right whale) is spotted, maintain a distance of at least 300 ft.

OBSERVE.3 Report sightings (not encountered, collided with, or injured by a project covered under 2020 SARBO) of the following species:

- North Atlantic Right whale: As defined in the North Atlantic Right Whale Conservation Plan (Appendix F) and the reporting requirements in the 2020 SARBO Section 2.9.
- Smalltooth sawfish: Report sightings to 1-844-SAWFISH or email Sawfish@MyFWC.com.

OBSERVE.4 Any collision(s) with an ESA-listed species must be immediately reported to the USACE and/or BOEM according to their internal protocol and to NMFS consistent with the reporting requirements in the 2020 SARBO Section 2.9. A vessel collision with an ESA-listed species is counted as take for the project.

In addition, reports of certain species shall also be reported as listed below. A link to the most current contact information will also be available at (**SERO.Dredge@noaa.gov**).

- Sea turtle take will also be reported to the appropriate state species representative (https://www.fisheries.noaa.gov/state-coordinators-sea-turtle-stranding-and-salvage-network).
- Smalltooth sawfish take will also be reported to 1-844-4SAWFISH or email Sawfish@MyFWC.com.

OBSERVE.5 Any collision with a marine mammal will be reported immediately to the Southeast Regional Marine Mammal Stranding hotline at 1-877-WHALE-HELP (1-877-942-5343) for guidance. This includes both ESA and non-ESA listed marine mammals.

2 PSO Credentials

All handling, tagging, and/or genetic sampling of ESA-listed species captured on projects covered under 2020 SARBO will be conducted by a PSO that meets the qualifications provided by NMFS.

PSO.1 Protected Species Training and Experience: PSOs selected to work on projects covered under 2020 SARBO will meet the following requirements:

- PSOs will meet the training and experience requirements outlined by NMFS. At the time of the completion of 2020 SARBO, PSO qualifications are confirmed by the NMFS Greater Atlantic Region Office, as defined on their website (https://www.fisheries.noaa.gov/new-england-mid-atlantic/careers-and-opportunities/protected-species-observers) for endangered species observers. A link to the current NMFS PSO qualifications will also be available on the NMFS SARBO webpage (**SERO.Dredge@noaa.gov**).
- PSOs will be trained and have experience to operate on the specific equipment they are aboard (e.g., hopper dredge, relocation trawler, G&G survey vessel). PSO will have training and/or experience to identify and handle all species that may occur in the geographic area of the project.
- PSO will be trained to safely install the specific tags being used and or collect genetic samples required under 2020 SARBO.
- ESA-listed species specific safe handling procedures, tagging procedures, and genetic sampling procedures must be followed, as outlined in these PSO PDCs. The most current procedures will be available on the NMFS SARBO webpage (**SERO.Dredge@noaa.gov**). The PSO must carry a copy of the PSO PDCs and all other applicable PDCs while on the vessel for easy reference.
- The 2020 SARBO serves as the authority for the PSO to handle, tag, and genetic sample ESA-listed species for those projects.

PSO.2 To minimize the risk of vessel collisions, a PSO trained in species observation is also responsible for monitoring for the presence of ESA-listed species when the vessel is in motion and must therefore have the training and experience needed to identifying ESA-listed species and marine mammals in their natural environment.

3 PSO Responsibilities

The Section outlines the responsibilities of a PSO working on a relocation trawler or hopper dredge. The PSO is also responsible for all other duties outlined in the PDCs of this appendix and those described in:

- The North Atlantic Right Whale Plan in Appendix F, if working in areas when and where North Atlantic right whale may be present as defined in Appendix F.
- The duties outlined in the hopper dredging PDCs in Appendix B, Section 3, if working on a hopper dredge.

Note: PSOs are also trained and may be responsible for monitoring for non-ESA-listed species including marine mammals protected under the Marine Mammal Protection Act. While the requirements outlined in the 2020 SARBO PDCs are limited to ESA-listed species, the PSO PDCs include guidance to minimize the risk of encounter with non-ESA listed marine mammals. The 2020 SARBO does not provide MMPA authorization.

3.1 PSO Guidance for handling ESA-listed species captured or observed injured or dead

The following PDCs describe how the PSO will handle ESA-listed species captured in hopper dredging and relocation trawling. If an ESA-listed species is observed injured or dead during other forms of dredging or material placement, this guidance also applies (e.g., observed during beach sand placement, in an upland disposal area, and while mechanical or cutterhead dredging).

PSO.3 PSOs observer coverage requirements are required to monitor for ESA-listed species as described below. PSOs on any project will not be assigned any other task (i.e., captain or other vessel crew position or task) while performing the role of PSO:

- Hopper dredging:
  - More than 1 PSO will be aboard the hopper dredge at all times.
  - The PSO on-duty is responsible for personally monitoring, handling, and reporting all captured ESA-listed species at all times when the hopper dredge is operating and follow the requirements of this Opinion including the hopper dredging PDCs in Appendix B, Section 3.
  - The PSOs will stand watch to detect ESA-listed species in the area and to alert the captain of their presence to avoid vessel collision whenever the vessel is moving. The on-duty PSO will only be responsible for standing watch and not performing other tasks such as inspecting or handling captures when the vessel is in motion.
- Relocation trawling: A PSO(s) will be aboard the trawling vessel at all times.
  - The PSO is responsible for all handling and reporting of ESA-listed species.
  - Trawling crew may assist in the removal of species from the nets and data recording only and the PSO is responsible for all tagging, genetic sampling, and assuring information reported is accurate.
  - All crew aboard the vessel, including the PSO, are responsible for monitoring for the presence of ESA-listed species in the area and reporting it to the vessel captain and PSO.

PSO.4 Reporting Captures of ESA-listed Species:

- Report to NMFS: All nonlethal captures and dead ESA-listed species observed or collected during a project covered under the 2020 SARBO will be recorded and reported to NMFS according to the procedures outlined in the 2020 SARBO Section 2.9. The captures will be recorded as follows:
  - Nonlethal take:
    - ESA-listed species captured and released back into the wild alive and healthy, will be considered nonlethal take.
    - If a sea turtle is entrained in a hopper dredge and is retrieved alive, the specialist such as a state sea turtle coordinator or sea turtle rehabilitation center specialist must be contacted to determine how the turtle should be handled (e.g., euthanized or taken to a rehabilitation facility). The take for a live turtle entrained in a hopper dredge is considered lethal until deemed healthy after an evaluation or rehabilitation and released back into the wild, then the take can be revised to be nonlethal.
    - If a sea turtle is captured in relocation trawling and is deemed unhealthy or injured and requires being sent to a specialist for further evaluation, the take is considered nonlethal, unless the species cannot be released back into the wild or dies, in which case the take must be updated to a lethal take.
  - Lethal take: All ESA-listed species that are captured that are determined to be fresh dead, will be considered lethal take associated with the project and counted under the total allowed take for the 2020 SARBO. This includes the capture of ESA-listed species in relocation trawling or found within the project area including material removal and material placement areas. An explanation of how to determine if a species is fresh dead or decomposed and how to handle and report the specimen is provided in PSO PDC Section 4 below.
  - Recovered dead: All ESA-listed species captured or observed in the project area that are decomposing will be considered a recovered specimen and will not be counted against the 2020 SARBO Incidental Take Statement. An explanation of how to determine if a species is fresh dead or decomposed and how to handle and report the specimen is provided in PSO PDC Section 4 below.
- Report captures to other agencies:
  - Sea turtles: All captures will be reported to the appropriate state species representative including live, fresh dead, and recovered dead (https://www.fisheries.noaa.gov/state-coordinators-sea-turtle-stranding-and-salvage-network).
  - Smalltooth sawfish take will be reported to 1-844-4SAWFISH or email Sawfish@MyFWC.com.

- Giant manta ray will be reported to manta.ray@noaa.gov.

PSO.5 Photo Documentation: Photograph all captured ESA-listed species for identification purposes and classify sex where applicable (e.g., sea turtles). In addition, take photographs of all injuries to ESA-listed species and provide a high resolution digital image with the take reporting forms as part of the reporting requirements outlined in the 2020 SARBO Section 2.9, as follows:

- Captures in relocation trawling that are not brought aboard the vessel or are released from the net will be photographed for identification purposes. Photographing should be done as quickly as possible to minimize the time the animal is out of the water and will not require manipulating the animal to improve the photograph.
- All injured, deceased, or otherwise debilitated sea turtles encountered during the course of dredging operations, whether intact, damaged, or partial remains, are thoroughly photographed.
- All surfaces should be clearly represented in the photos with both wide vantage and close-up images that portray any injuries and postmortem condition (if deceased).
- Minimally, this includes multiple images of the dorsal (top) and ventral (bottom) aspects of each specimen taken from different angles and perspectives.
- An identification placard and scale should appear in the images but should not obscure the specimen, injury, or specific area of interest. The identification placard will include the location of capture, date, time, and species. In addition, the time settings on the camera should be current so that the time stamp within the photo metadata is accurate.
- For any live capture that is injured or otherwise debilitated and will be taken to a rehabilitation facility, photographs can be delayed in order to minimize stress and risk of further injury prior to veterinary examination.
- For deceased specimens, photos will be taken within 2 hours following discovery so that postmortem state in the images accurately portrays the condition at the time of discovery.

PSO.6 Written Documentation: Document all relevant details of the capture according to the reporting requirements in the 2020 SARBO, Section 2.9 (e.g., species, size, sex, condition upon release, location of capture, and time of capture) that can be observed or measured by the PSO without causing harm to the animal.

PSO.7 Tagging: Nonlethal captures of ESA-listed species captured by projects covered under the 2020 SARBO will be tagged according to the following requirements. Tagging requirements only apply to those ESA-listed species that are brought aboard a relocation trawler (PSO PDC Section 3) or those captured and ultimately released alive from a hopper dredge after being evaluated by a specialist and/or rehabilitated.

- Scanning: All ESA-listed species (live and dead) and/or species parts captured by a hopper dredge or brought aboard a relocation trawler will be scanned for passive integrated transponder (PIT) tags to determine if the animal has been previously tagged. The presence of any external tags (e.g., flipper tags, dart tags) will also be noted. All previous tag numbers must be recorded and reported on the appropriate forms outlined for each species in PSO PDC Sections 5-9 below.

- Tagging: All ESA-listed species captured alive and in good health by a hopper dredge or brought aboard a relocation trawler that are scanned and lack a previous pit tag, will be PIT tagged according to the specific species procedures identified in PDC PSO.7. Additional external tags (e.g., flipper tags) are optional. The cost associated with tagging is the responsibility of the federal action agency overseeing the project (i.e., USACE or BOEM) or the company awarded the contract.

PSO.8 Genetic Sampling: All nonlethal and lethal captures ESA-listed species captured by projects covered under the 2020 SARBO will be have genetic samples taken except:

- Live ESA-listed species that are not brought aboard a relocation trawler (PSO PDC Section 3.2).

- Any leatherback sea turtles, even if brought aboard the vessel to untangle and safely release.

- Any shortnose sturgeon.

- If the PSO believes that collecting a sample would imperil human or animal safety. The rationale for this decision will be recorded on the species observation form and available digitally as part of the reporting requirements outlined in the 2020 SARBO Section 2.9.

PSO.9 Genetic samples will be collected according to the handling procedures defined for each species in the PSO PDCs Section 5-9 below.

- A tissue sample will be collected from any dead ESA-listed species. If multiple dead animal parts are found, a sample will be collected from all parts that are not connected to one another regardless of whether the tissues are believed to be from the same turtle. For example, if part of a sea turtle flipper and a detached head are found at the same time, a sample from each part will be collected for genetic analysis.

- All genetic samples will be preserved in RNAlater™ preservative. Once the sample is in buffer solution, refrigeration/freezing is not required, but care should be taken not to expose the sample to excessive heat or sunlight. Label each sample with the animal's unique identification number (PIT tag number). Since giant mantas will not be pit tagged, label any samples collected with the date, project name, and species name. Do not use glass vials; a 2 millimeter screw top plastic vial is preferred (e.g., MidWest Scientific AVFS2002 and AVC100N). Gently shake the sample to ensure the solution covers the entire sample.

- Genetic samples will be mailed to the addresses listed below with information provided in the container stating the sample was collected under the 2020 SARBO (Project name, SARBO SER-2008-05934). Package the genetic samples with an absorbent material within a double-sealed container (e.g., zip lock bag). If more than 1 sample is being sent to an address, package all of the samples together. The cost associated with taking the sample and delivering it to the appropriate entity listed below is the responsibility of the federal action agency overseeing the project (i.e., USACE or BOEM) or the company awarded the contract.
  - Sea turtles: Sea Turtle Program NOAA Southeast Fisheries Science Center Attn: Lisa Belskis, 75 Virginia Beach Drive, Miami, Florida 33149. Contact number: 305-361-4212 Lisa.Belskis@noaa.gov
  - Sturgeon: Geological Survey Leetown Science Center, Attention Robin Johnson, Aquatic Ecology Branch, 11649 Leetown Road, Kearneysville, West Virginia 25430.
  - Elasmobranchs: NOAA Southeast Fisheries Science Center, Attention Dr. John Carlson, National Marine Fisheries Service, Panama City Laboratory, 3500 Delwood Beach Rd, Panama City, Florida, 32408).

PSO.10 Atlantic Sturgeon Genetic Sampling Testing Requirements under the 2020 SARBO:

- The USACE, BOEM, or entity designated by the USACE or BOEM are responsible for the cost to analyze/test genetic samples from Atlantic sturgeon captured to determine the DPS of Atlantic sturgeon captured (live and dead captures).

- Atlantic sturgeon genetic samples will be recorded on and submitted with the Sturgeon Genetic Sample Submission spreadsheet available on the NMFS dredging webpage https://www.fisheries.noaa.gov/content/southeast-dredging. This form should indicate in the "comment field" if the Atlantic sturgeon was previously PIT tagged. A copy of that reporting spreadsheet will also be sent to NMFS (takereport.nmfsser@noaa.gov), the genetic sampling address in PDC PSO.8 above along with the sample, and to mike_mangold@fws.gov.

3.2 PSO Guidance on Relocation Trawling

The following PDCs describe how the PSO will handle ESA-listed species captured during relocation trawling including a flow chart summarizing how to handle different species and text describing the general handling guidance, the order to release species if multiple ESA-listed species are captured in trawling, and where they should be released. Trawling within the range of ESA-listed corals is not covered under this Opinion (Appendix B, Section 3.5).

Table 59. PSO Handling Guidance

Species and handling protocol	Handling priority for multiple captures	Required to bring aboard (Y/N)	Directly measure all required data (SARBO Section 2.9)[F]	Estimate all required data (SARBO Section 2.9)	Photograph (PDC PSO.5)	Tagging and Genetic Sampling (PDC PSO.7-10)	Relocate
Smalltooth sawfish PSO PDC Section 7	1	A, C	No	Yes	Yes	No	No
Sharks PSO PDC Section 9	2	A	No	Yes	Yes	No	No
Giant manta ray PSO PDC Section 8	3	A, C	No	Yes	Yes	No	No
Leatherback sea turtle PSO PDC Section 5	4	A	No	Yes	Yes	No	No
Sturgeon PSO PDC Section 6	5	B, D, F	Yes	No	Yes	Yes	G
Green, hawksbill, Kemp's ridley, loggerhead sea turtles PSO PDC Section 5	6	B, E	Yes	No	Yes	Yes	F

A. Animals will not be brought aboard and will remain and be released from net while still in water. If necessary, cut the net to expedite release.
B. Animals will be brought aboard, except if the PSO directs removal from net to protect the safety of the animal or crew (e.g., turtle in net with large shark).
C. If juvenile manta rays or smalltooth sawfish need to be brought aboard to safely disentangle, only allowed if animal is small enough to be picked up by crew and released according to PSO handling guidance
D. Sturgeon will be brought aboard and place in holding tank. Must release within 30 minutes (20 minute if tank unavailable), even if not relocated.
E. Turtle will be kept cool, wet, and kept in a safe area such as a kiddie pool to contain for safe transportation to the relocation site. If sick, injured, or requiring resuscitation, see PSO PDC Section 5 for guidance.
F. Animals brought aboard will be measured and data collected as quick as possible to return them to the water safely.
G. Relocate according to guidance in PDC PSO.11-13.

PSO.11 <u>Marine Relocation Trawling</u>: Relocation trawling is authorized in the marine environment as a measure to minimize lethal take of ESA-listed species.

- Sea turtles (with the exception of leatherback sea turtles) and sturgeon will be relocated 3-5 miles from the dredge project, if relocation can be done safely, according to the guidance in PSO PDC Section 5.
- The PSO will determine the appropriate release site based on the species captured and surrounding habitat.

PSO.12 <u>Estuarine Relocation Trawling</u>: Relocation trawling is authorized in the estuarine environment as a measure to minimize lethal take of ESA-listed species.

- For the purposes of relocation trawling authorized in the 2020 SARBO, the estuarine environment consists of bays, harbors, estuaries, or other semi-confined areas inland of the COLREGS Demarcation Line, but outside of a river. The start of a river is not defined and varies by location and should be determined by best professional judgment. When in doubt, NMFS may be contacted for clarification.
- Atlantic sturgeon 4 ft (1.2 m) total length or larger may be relocated to any location, including marine waters. Atlantic sturgeon less than 4ft total length and all shortnose sturgeon must be relocated within the estuary where it was captured.
- The PSO will determine the appropriate release site based on the species captured and surrounding habitat.

PSO.13 <u>Riverine Relocation Trawling</u>: Relocation trawling is not authorized within rivers, as noted in PDC PSO.12 above, the start of a river is not defined and should be determined by best professional judgment.

PSO.14 For relocation trawling:

- If any marine mammals, or aggregation of any other species not targeted for relocation trawling (e.g., fever of rays or school of fish) are sighted prior to deploying the nets and believed to be at-risk of interaction (e.g. moving in the direction of the vessel/gear and moms/calves close to the gear), gear deployment should be delayed until the animal(s) are no longer at-risk or have left the area of their own volition.
- During relocation trawling, the PSO and vessel staff will monitor for species presence at all times. Gear will be immediately retrieved if marine mammals or other species not targeted for relocation trawling are believed to be captured/entangled or at-risk of capture/entanglement. Operations may resume when interaction with these species is deemed unlikely, based on best professional judgment and through coordination with the PSO onboard.
- If a non ESA-listed marine mammal is injured or captured during relocation trawling, we recommend that trawling activities cease if other marine mammals may be in the area that are at risk of capture until provided

guidance on how to proceed by NMFS or the marine mammal stranding staff. The 2020 SARBO does not consider effects to non-ESA-listed marine mammal species.

PSO.15 Relocation trawling handling and training:

- The PSO will train all crew members on the vessel how to safely handle and remove animals from the net and record tow capture data.
    - o Training will occur with each new crew before heading out to begin work (e.g., if the crew will be at sea for 3 weeks before rotating staff, the training will be done at the beginning of the 3-week period, even for crews that have worked together before).
    - o ESA and non-ESA listed animals captured may be removed from the net by crew other than the PSO, if trained by the PSO on proper handling and release techniques to minimize the risk of harm to these animals.
    - o All ESA-listed species tagging, and genetic sampling will be performed by the PSO. Other crew members may assist with data collection, which be checked by the PSO for accuracy before reporting.
- All crew members will have easy access to equipment used to untangle animals from the net or to cut the nets to free the animal including knives, line-cutting poles, long-handled dehookers, and/or boat hooks.
- The nets will be checked during every tow for the presence of ESA-listed species. This may require pulling the tail end of the net to the boat to confirm nothing has been captured.
- For all species, ensure the vessel is in neutral and release animal over the side, head first.

4 Handling and Reporting Dead ESA-listed Species

All dead ESA-listed species collected within the construction area or by equipment used on a project covered under the 2020 SARBO, will be handled and recorded as described in the PSO PDCs and 2020 SARBO Section 2.9.

PSO.16 Dead ESA-listed species collected within the area of work will be rated as fresh dead or decomposed and documented as described in PSO PDC.4. The determination of a specimen's condition (fresh dead or decomposed) is as follows:

- Decomposed specimens are those that exhibit obvious bloating (expansion of the body or tissues by putrefactive gases); detachment of skin upon handling; or liquefaction of organs and tissues. Examples of decomposition in sea turtles are provided in Figure 75 below. Note: foul odor alone is not considered definitive evidence of decomposition.
- If it is not clear whether the specimen is fresh dead or decomposed, the specimen will be retained for further examination by an individual that has demonstrated expertise in sea turtle necropsy and forensic pathology. Such examinations typically include complete gross examination and selective histopathology, depending on postmortem condition. Individuals that will conduct examinations should be identified prior to the onset of dredging operations along with the necessary logistical planning for transportation and storage needs. The associated stranding coordinator for the state or region of the operation may be able to advise or assist in this regard as such needs are regularly required during stranding response. NMFS retains the right to review evidence or seek the opinion of an expert if a take determined to be decomposed should have been listed as fresh take and take associated with the project.

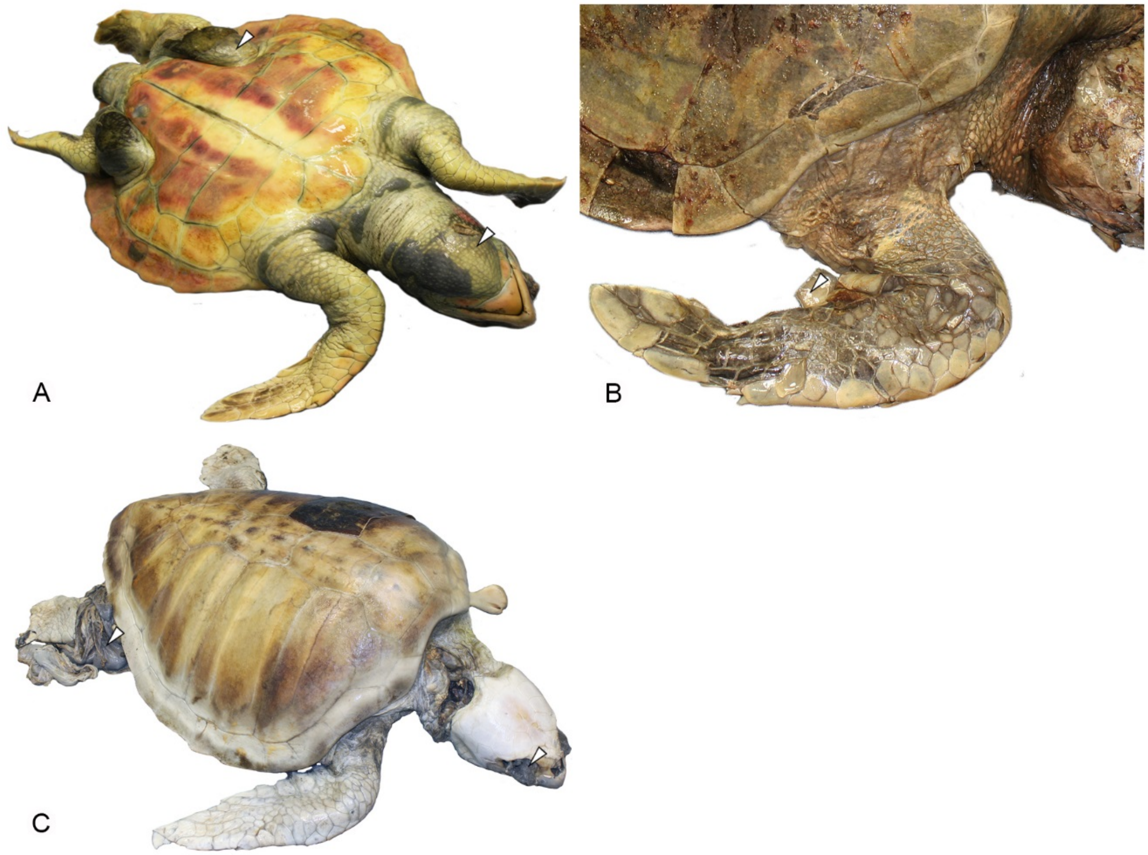


Figure 75. Examples of obvious signs of decomposition.
(A) Bloating expands the loose skin around the flippers and neck. (B) The skin starts to detach in sheets. (C) Soft tissues beginning to fall apart and easily tear when handled.

PSO.18 Dead ESA-listed species and species parts that need further examination by a specialist to determine the cause of death will be refrigerated, iced, or frozen as soon as possible, (must be iced or frozen no more than 2 hours from discovery). The timeline from discovery to transfer for examination, including ambient temperature, must be thoroughly documented. Whether the carcass/part is refrigerated or frozen will depend on predetermined logistical parameters for a given project. In general, a carcass/part may be kept refrigerated or iced, but not frozen if it will be examined within 48 hours. Remains may be frozen if examination will be delayed or maintaining refrigeration is not possible for any reason.

- Dead turtles: Follow the protocol outlined on the *Protocol for Collecting Tissue From Dead Turtles for Genetic Analysis* (https://dqm.usace.army.mil/odess/documents/geneticsampleprotocol.pdf). If a revised document is released, the PSO is required to follow the revised protocols. This document and any revisions will also be available on the NMFS dredging webpage (https://www.fisheries.noaa.gov/content/southeast-dredging).

- Dead sturgeon specimens will be reported to 1-844-STURG911 (1-844-788-7491) and via the Sturgeon Salvage Form (available on our Dredging website at: https://www.fisheries.noaa.gov/content/southeast-dredging). In addition, a fin clip and a fin ray will be collected in accordance with the Genetic Sampling Collection Requirements described in PSO PDC Section 3.1 using the genetic submission form (available on our Dredging website at: https://www.fisheries.noaa.gov/content/southeast-dredging).
- Dead elasmobranchs specimens will be stored as described in PDC PSO.16 until advised how to dispose of or provide to Dr. John Carlson, NOAA Fisheries, Panama City Laboratory at 1-850-234-6541 x 221. Dead smalltooth sawfish will also be reported to 1-844-4SAWFISH (1-844-472-93474).

5.1 Identification

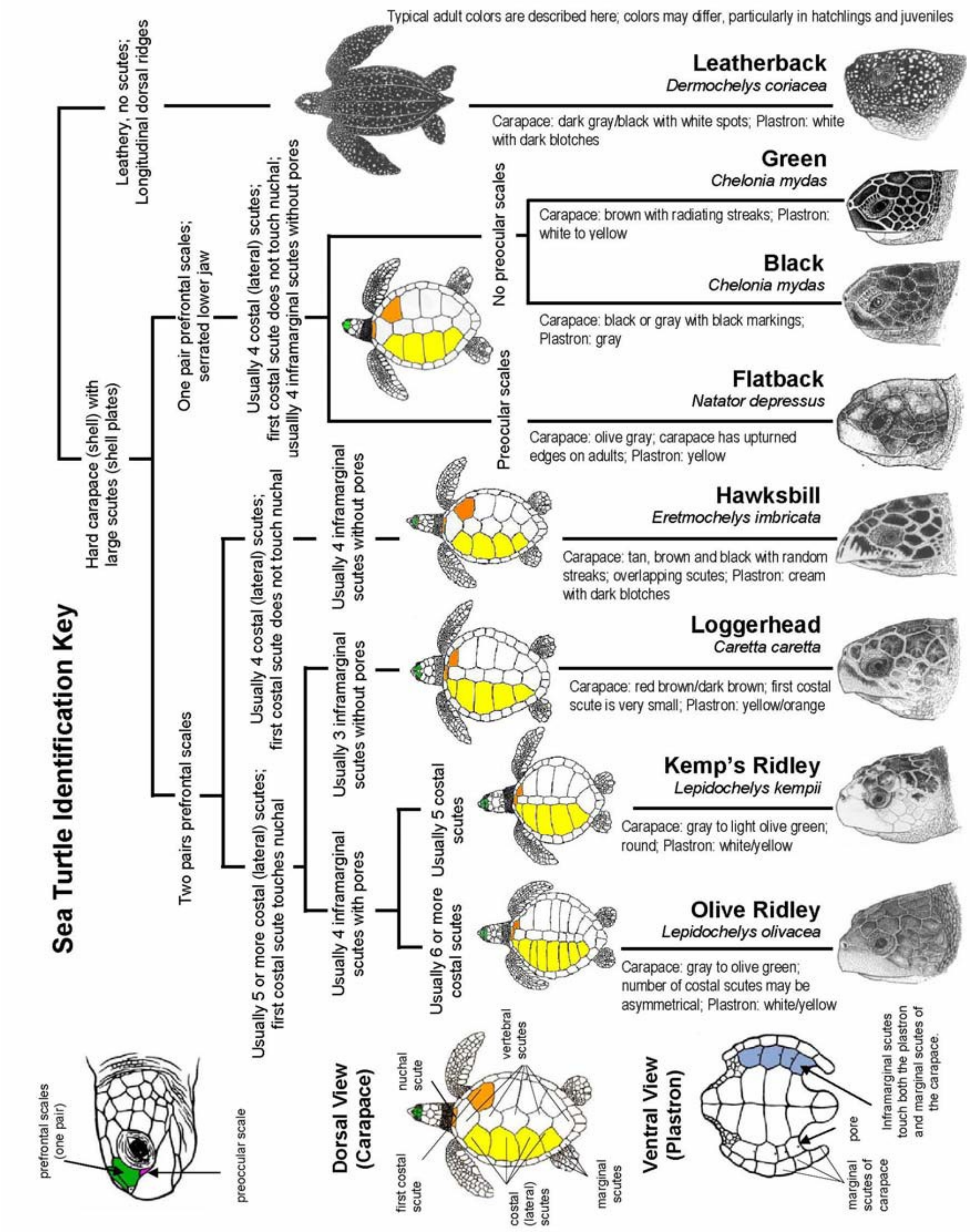


Figure 76. Sea Turtle Identification Key Image from the Southeast Fisheries Science Center Sea Turtle Research Techniques Manual, updated January 2013 (NOAA Technical Memorandum NMFS-SEFSC-579, https://repository.library.noaa.gov/view/noaa/3626)(NMFS 2008)

5.2 Handling

- Sick or injured sea turtles will be evaluated by a specialist to determine the best course of action including euthanizing animals that are severely injured or rehabilitating sea turtles before releasing them back in to the wild.
- A specialist trained to evaluate sea turtles and a sea turtle rehabilitation center will be identified prior to starting a project. Directions of how sick or injured sea turtles will be transported for an evaluation or rehabilitation will be provided to the PSO and dredging or trawling staff. NMFS will assist with identifying specialist and rehabilitation centers, if needed.

5.3 Relocating

- Do not relocate leatherback sea turtles. Release them immediately, as described in PSO Section 3.2 above.
- Green, Kemp's ridley, loggerhead, and hawksbill sea turtles will be relocated and released not less than 3 nautical miles (nmi) from the dredge site, unless sick or injured. If 2 or more released turtles are later recaptured, subsequent turtle captures will be released not less than 5 nmi away. If it can be done safely and without injury to the turtle, turtles may be transferred onto another vessel for transport to the release area to enable the relocation trawler to keep sweeping the dredge site without interruption. These turtles will be kept no longer than 12 hours prior to release. The area in which a turtle will be relocated is determined by the PSO.

5.4 Data Recording

- Record the carapace length and width (straight and curved measurements), plastron length and width, head width, and sex (if possible).
  - Follow the protocol outlined in the *Southeast Fisheries Science Center Sea Turtle Observer Manual*, updated January 2013 (NOAA Technical Memorandum NMFS-SEFSC-589, https://repository.library.noaa.gov/view/noaa/4392). Additional, specific handling techniques are required when handling turtles with fibropapilloma tumors. If a revised document is released, the PSO is required to follow the revised protocols. This document and any revisions will also be available on the NMFS dredge webpage (Https://www.fisheries.noaa.gov/content/southeast-dredging).

5.5 Tagging and Genetic Sampling

- Follow the protocol outlined in the *Southeast Fisheries Science Center Sea Turtle Observer Manual*, Updated January 2013 (NOAA Technical Memorandum NMFS-SEFSC-589, https://repository.library.noaa.gov/view/noaa/4392). If a revised document is released, the PSO is required to follow the revised protocols. This document and any revisions will also be available on the NMFS dredge webpage (Https://www.fisheries.noaa.gov/content/southeast-dredging).

- Tagging and genetic sampling of leatherback sea turtles is not required under this Opinion and priority should be given to quickly and safely release the animals due to the sensitivity of these animals to being handled.

5.6 Resuscitation

- Follow the protocol outlined in the *Southeast Fisheries Science Center Sea Turtle Observer Manual*, Updated January 2013 (NOAA Technical Memorandum NMFS-SEFSC-589, https://repository.library.noaa.gov/view/noaa/4392). If a revised document is released, the PSO is required to follow the revised protocols. This document and any revisions will also be available on the NMFS dredge webpage (https://www.fisheries.noaa.gov/content/southeast-dredging).

6 Sturgeon Handling, Tagging and Genetic Sampling Protocol for Relocation Trawling

6.1 Identification

Shortnose sturgeon are similar in appearance to Atlantic sturgeon, but can be distinguished by their color, generally smaller size, wider mouth, and smaller snout shape (Figure 77).

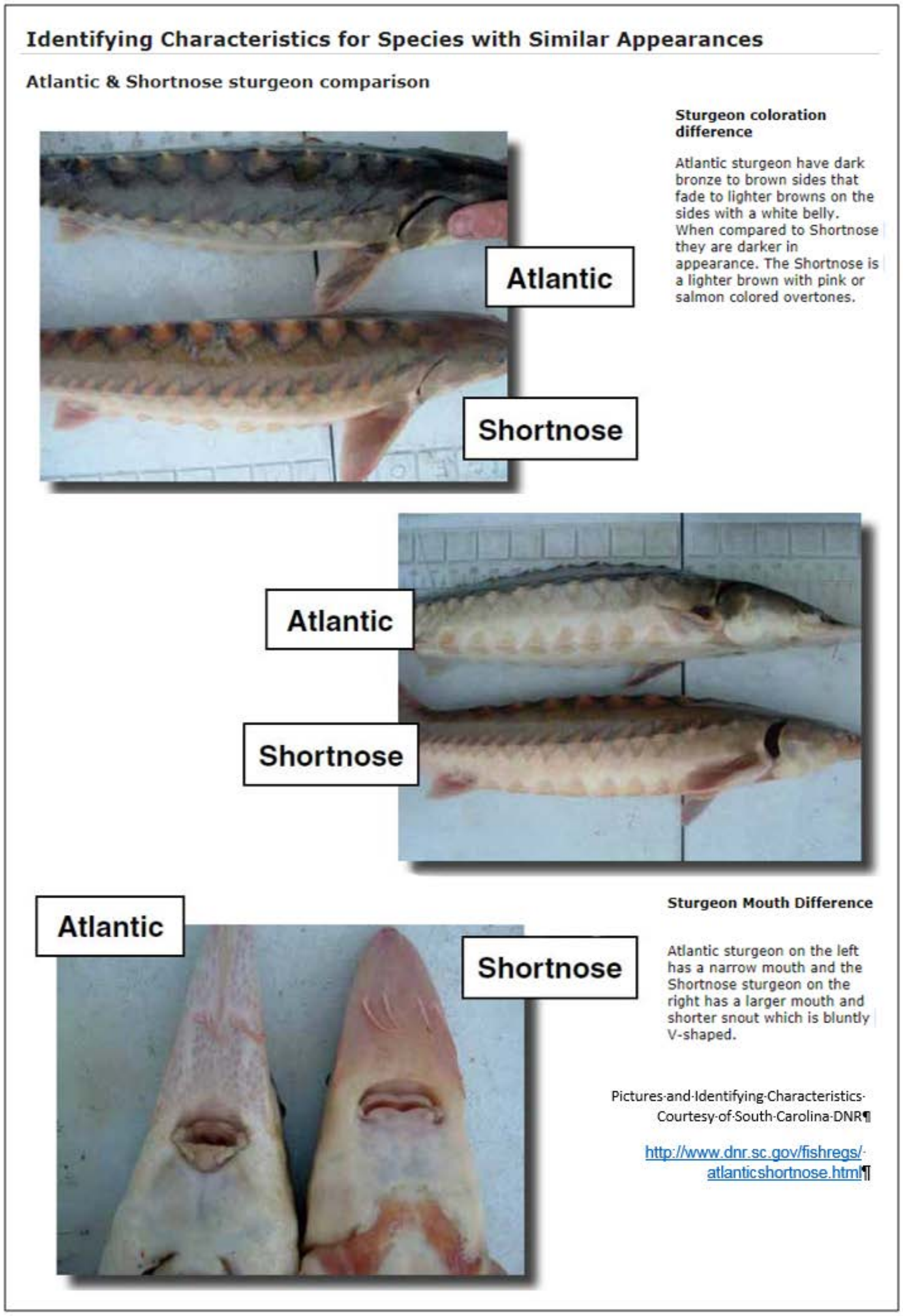


Figure 77. Atlantic and shortnose sturgeon species identification guide

6.2 Handling

- In areas where sturgeon are likely to be captured, the relocation trawler will have a sturgeon holding tank onboard that will be used during handling and relocation. The tank will be of sufficient size and shape to safely hold the size sturgeon that may be captured in the area and water from the surrounding environment must be continuously circulated and exchanged with the water in the tank to ensure it remains the proper temperature and properly oxygenated.
- If a sturgeon cannot be held in a holding tank during handling, the sturgeon will be kept wet at all times using water from the surrounding environment. Other options for holding the sturgeon while handling it include using a net pen/basket floating placed next to the vessel.
- While moving the animal or removing it from gear, covering its eyes with a wet towel may help calm it.
- All handling procedures (i.e., measuring, PIT tagging, photographing, and tissue sampling) should be completed as quickly as possible, and should not exceed 20 minutes from when the sturgeon is first brought on board the vessel or 30 minutes if placed in a holding tank (see above). Handling procedures should be prioritized in the following order: (1) collect a tissue sample (see procedure described below); (2) scan for existing PIT tags, apply new PIT tag if no pre-existing PIT tag is found; (3) measure the animal; (4) photograph the animal. If all of the handling procedures cannot be completed within 20 minutes, the animal should be returned to the water; indicate which procedures were not completed when reporting the incidental take to NMFS.

6.3 Relocating

- Sturgeon will be released immediately after capture, away from the dredge site or into already dredged areas, unless the relocation trawler is equipped with a suitable well-aerated seawater holding tank, container, trough, or pool where a maximum of a single fish may be held for not longer than 30 minutes before it must be released or relocated away from the dredge site. The area in which a sturgeon will be relocated is provided in PDCs PSO.11-13 with the exact location determined by the PSO.

6.4 Data Recording

- Length measurements for all sturgeon should be taken as a straight line measurement from the snout to the fork in the tail (i.e., fork length), and as a straight line measurement from the snout to the tip of the tail (i.e., total length) (Figure 78). Do not measure the curve of the animal’s body.

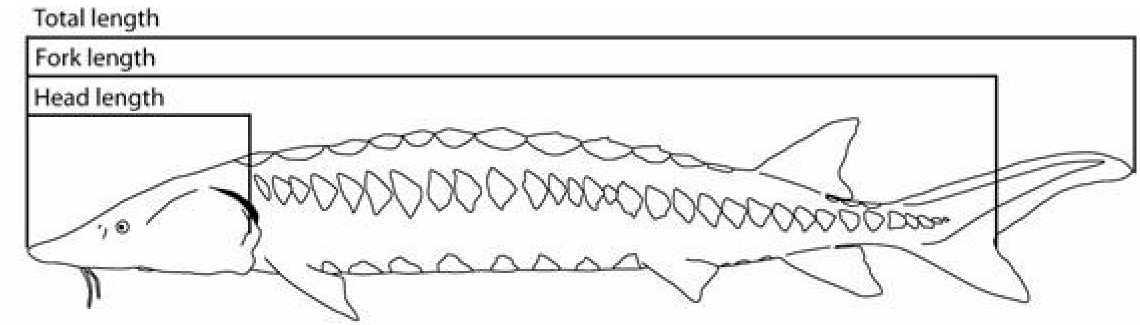


Figure 78. Diagram of different types of measurements for sturgeon.
Drawings by Eric Hilton, Virginia Institute of Marine Science in Mohead and Kahn 2010 (Kahn and Mohead 2010).

6.5 Tagging

- Every sturgeon should be scanned for PIT tags along its entire body surface ensuring it has not been previously tagged. The PIT tag readers must be able to read both 125 kHz and 134 kHz tags.
- Animals without an existing PIT tag will receive one that operates at a frequency of 134.2 kHz.
- Sturgeon smaller than 250mm will not be PIT tagged. Sturgeon measuring 250-350 mm total length will only be tagged with 8mm PIT tags. Sturgeon 350 mm or greater will receive standard sized PIT tags (e.g., 11 or 14 mm).
- PIT tags should be implanted to the left of the spine immediately anterior to the dorsal fin, and posterior to the dorsal scutes (Figure 79). This positioning optimizes the PIT tag's readability over the animal's lifetime. If necessary, to ensure tag retention and prevent harm or mortality to small juvenile sturgeon of all species, the PIT tag can also be inserted at the widest dorsal position just to the left of the 4th dorsal scute.
- Scan the newly implanted tag following insertion to ensure it is readable before the animal is released. If the tag is not readable, 1 additional tag should be implanted on the opposite side following the same procedure, if doing so will not jeopardize the safety of the animal.

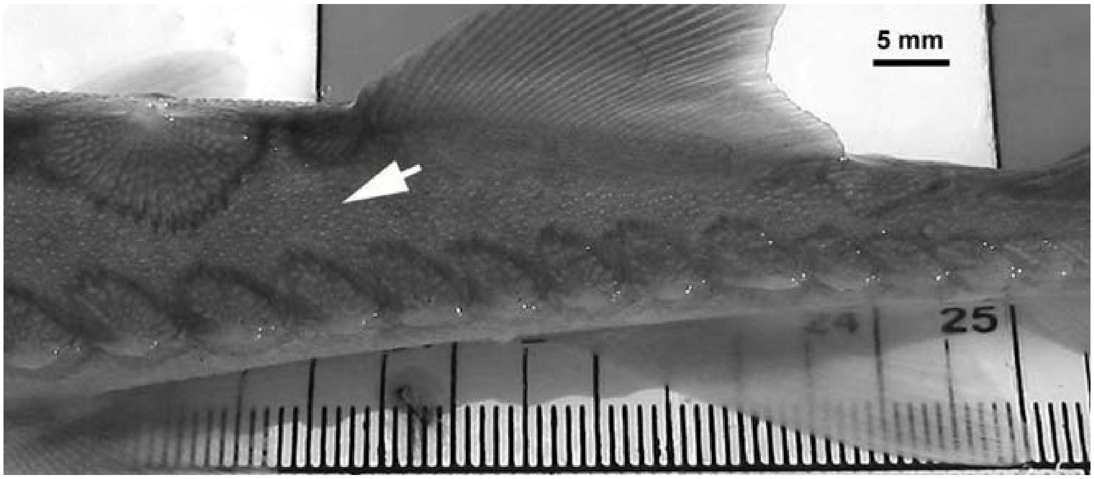


Figure 79. Standardized Location for PIT Tagging all Gulf, Atlantic, and shortnose sturgeon (Photo Credit: J. Henne, USFWS)

6.6 Genetic Sampling

- Tissue samples should be a small (1.0 cm^2) fin clip collected from soft pelvic fin tissue. Use a knife, scalpel, or scissors that has been thoroughly cleaned and wiped with alcohol.
- Collected genetic samples must be stored in accordance with the requirements described in PSO PDC Section 3 above.

7 Smalltooth Sawfish Handling, Tagging and Genetic Sampling Protocol for Relocation Trawling

7.1 Identification

- The smalltooth sawfish is distinguished by the 22 to 29 teeth located on each side of the rostrum and the lack of a bottom lobe on the caudal (tail) fin (Figure 80).

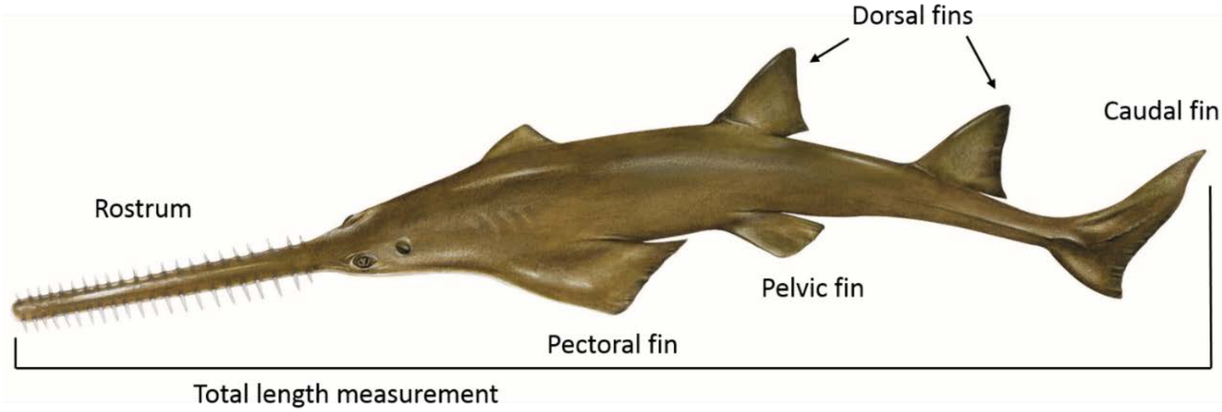


Figure 80. Image of a smalltooth sawfish.

7.2 Handling

- Attempt to release the sawfish directly from the net by pulling the net alongside the vessel and cutting the net sections that are entangled. Keep the sawfish in the water as much as possible during this process.
- Only bring the sawfish aboard the vessel if absolutely necessary to free it from the net. Larger animals should never be brought aboard due to the difficulty of returning them safely to the water. If necessary to bring a smaller animal aboard to free it from the net, make sure to keep sawfish wet and work quickly to get it safely back in the water. Smaller sawfish can be returned to the water by 2 people with the first person grasping the animal at the base of the rostrum with one hand and supporting the mid-section with the other. The second person can grasp the animal at the base of the tail and support the mid-section with the other hand.
- Use caution when near the rostrum as it can sweep side-to-side and cause injury during handling.
- Do not grab the sawfish by the spiracles (holes on the top of the head).

7.3 Relocating

- Do not relocate smalltooth sawfish. It is more important to release them as soon as possible as described above.

7.4 Data Recording

- Record the total length of the sawfish and the number of teeth on each side of the rostrum (saw). Estimate the length and number of teeth based on the photo taken of the sawfish in the net, if necessary.

7.5 Tagging and Genetic Sampling

- Tagging and genetic sampling of smalltooth sawfish is not required under this Opinion and priority should be given to quickly and safely release the animals due to the sensitivity of these animals to being handled.
- If the sawfish is brought aboard the vessel to untangle it from the net, scan the sawfish for a PIT tag generally located in the muscles at the base of the first or second dorsal fin.
- If no PIT tags are found, implant a BIOMARK HPT 12 PIT tag (12.5 mm in length, 134.2 kHz) under the skin directly adjacent to the second dorsal fin.
- Scan the newly implanted tag following insertion to ensure it is readable before the animal is released. If the tag is not readable, 1 additional tag should be implanted on the opposite side following the same procedure, if doing so will not jeopardize the safety of the animal.
- If a tissue sample is taken, it should consist of a small (1.0 cm^2) fin clip taken from the posterior edge of one of the pelvic fins. Use a thoroughly cleaned (wiped with alcohol) knife, scalpel, or scissors to collect the sample.
- Collected genetic samples must be stored in accordance with the requirements described in PSO PDC Section 3 above.

7.6 Additional Resources for Review

- Sawfish Fact Sheet (https://www.fisheries.noaa.gov/resource/educational-materials/endangered-smalltooth-sawfish-fact-sheet)
- Sawfish Handling, Release, and Reporting Procedures (https://www.fisheries.noaa.gov/resource/educational-materials/endangered-sawfish-handling-release-and-reporting-procedures)

8 Giant Manta Handling Data Recording, and Genetic Sampling Protocol for Relocation Trawling

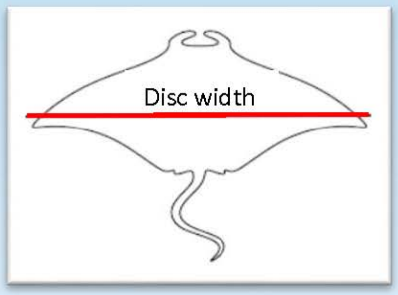
8.1 Identification

NOAA FISHERIES

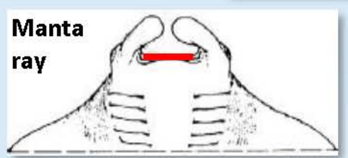
Mobula Ray Identification Guide For Fisheries Observers

Purpose: This guide is intended to assist fishery observers in the visual identification of the giant manta ray and several devil ray species that occur in the Southeast and Mid-Atlantic.

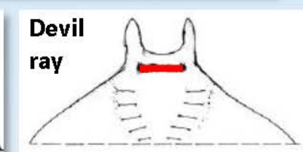
General Observations: The size, coloring patterns, and a few morphological differences can be used to distinguish between species.



- Giant manta rays are larger than devil rays. Measurements should be taken by estimating the distance over their wingspan ["Disc Width" (DW)].
- Giant manta rays have a terminal mouth (i.e., mouth points straight forward, in front of the head); Devil rays have a sub-terminal mouth (i.e., mouth beneath the head).



Terminal mouth



Sub-terminal mouth

Manta birostris

Common Names: Giant Manta Ray, Oceanic Manta Ray

Status: U.S.: Listed as *Threatened* under Endangered Species Act.

Size: Up to 700 cm DW; appx. 200 cm DW at birth.

Dorsal Coloration: Black with distinct white patches creating a T-shaped shoulder pattern.

Ventral Coloration: White with dark spots; spots rarely found between gill slits. Dark shading along the posterior edges of the pectoral fins.

Photo credit: Joshua Stewart

NOAA Fisheries, Southeast Region, Protected Resources Division

Mobula mobular

Common Names: Giant Devil Ray, Spinetail Devil Ray

Status: U.S.: Not listed. International Union for Conservation of Nature (IUCN): *Endangered*

Size: Up to 520 cm DW

Dorsal Coloration: Predominantly dark gray; with a black (crescent shape) stripe that runs from side to side on upper shoulders. White tip on the dorsal fin.

Ventral Coloration: White.

Mobula tarapacana

Common Names: Chilean Devil Ray, Sicklefin Devil Ray, Box Ray

Status: U.S.: Not listed. IUCN: *Vulnerable*

Size: Up to 340 cm DW

Dorsal Coloration: Golden brown to olive green.

Ventral Coloration: Predominately white with gray shading along the posterior margin of pectoral fins.

Mobula hypostoma

Common Names: Atlantic Devil Ray, Lesser Devil Ray

Status: U.S.: Not listed. IUCN: *Data Deficient*

Size: Up to 120 cm DW

Dorsal Coloration: Variable, brown, gray to black. Sometimes have a dark gray/black stripe that runs from side to side on the "neck" right behind the eyes.

Ventral Coloration: White.

Figure 81. Mobula Ray Identification Guide

8.2 Handling

- Removing the giant manta ray from the water can increase the likelihood of injuries, mostly due to the crushing the animal's organs due to the weight of gravity.
- If a manta ray needs to be brought aboard, support the ray's weight by at least 2 points (i.e. one point of contact being the midsection, and the other being the posterior end of the body) or preferably have 2 or 3 people carry the ray by the sides of each wing.
- Follow the safe handling guidance: https://www.fisheries.noaa.gov/webdam/download/91927887

8.3 Relocating

- Do not relocate giant manta rays. Release them immediately, as described above.

8.4 Data recording

- Record the total disc width from wing tip to wing tip, as shown in the *Mobula Ray Identification Guide for Fisheries Observers* (Figure 81). Estimate the disc width, if released directly from the net and not brought aboard the vessel.
- Photograph animal. Manta's have unique spot patterns on the ventral side used for identification so photograph as much of the animal as possible without flipping or manipulating the animal.

8.5 Tagging and Genetic Sampling

- Tagging and genetic sampling of giant manta rays is not required under this Opinion and priority should be given to quickly and safely release the animals due to the sensitivity of these animals to being handled. Tagging of giant mantas is not recommended under the 2020 SARBO unless it is part of cooperative research with NMFS.
- If a tissue samples is taken, it should be a small tissue (1.0 cm2) fin clip taken from dorsal fin or posterior edge of pectoral fin. Use a knife, scalpel, or scissors that has been thoroughly cleaned and wiped with alcohol.
- Collected genetic samples must be stored in accordance with the requirements described in PSO PDC Section 3 above.

8.6 Additional Resources for Review

- Giant manta ray safe handling guidelines, https://repository.library.noaa.gov/view/noaa/22926 (Carlson et al. 2019).
- The giant manta ray can be visually distinguished from other rays by size, coloring, and a few morphological differences, as shown in *Mobula Ray Identification Guide for Fisheries Observers* (Figure 81).

9 Shark Handling, Tagging and Genetic Sampling Protocol for Relocation Trawling

9.1 Identification

Scalloped hammerhead and oceanic whitetip shark are ESA-listed species occurring within the action area. However, oceanic whitetip shark are a deep water (pelagic) species that are not expected to be captured during relocation trawling. Scalloped hammerhead shark may be encountered during relocation trawling, but are only protected under the ESA if they are a part of the Central and Southwest Atlantic DPS, which would only expected in the U.S. Caribbean (79 FR 38242). Scalloped hammerhead sharks encountered outside of the U.S. Caribbean are not protected under the ESA, but are still expected to be handled according to the PSO guidance in this Appendix.

Identification of both scalloped hammerhead and oceanic whitetip shark are provided on the placard used for the *Shark Identification and Federal Regulations for the Recreational Fishery of the U.S. Atlantic, Gulf of Mexico, and Caribbean* shown in Figure 82 along with identification guidance for other sharks that may be encounter that are not ESA-listed. Safe handling practices outlined in this section will be followed regardless of the ESA-listing status of the shark encountered.

9.2 Handling

- Large sharks should be released directly from the net into the water and not brought aboard the vessel.
- If sharks must be brought aboard to safely remove them from the net, cut the net quickly and release them back to the water. If necessary to bring a smaller animal aboard to free it from the net, make sure to keep shark wet and work quickly to get it safely back in the water. Smaller sharks can be returned to the water by grasping the animal under the jaw and ensuring the jaw is closed. Depending on the size of the shark, this may require 2 hands to hold the jaw closed while a second crew member helps to carry the shark back to the water.
- Sharks are reported to frequently chew through a portion of the net and are retrieved trapped in the net at the gills. In instances such as this, the net will be quickly cut and the shark removed.
- Do not pull sharks free or carry them by the gills.

9.3 Relocating

- Do not relocate sharks. It is more important to release them as soon as possible and described above.

9.4 Data Recording

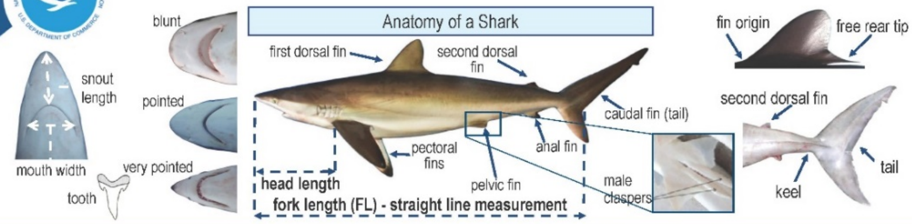
- Record the total length of the shark either by measuring the shark if brought aboard or by estimating the length based on the photo taken of the shark in the net, if necessary.

9.5 Tagging and Genetic Sampling

- Tagging and genetic sampling of sharks is not required under this Opinion and priority should be given to quickly and safely release the animals due to the sensitivity of these animals to being handled.

Shark Identification and Federal Regulations

for the Recreational Fishery of the U.S. Atlantic, Gulf of Mexico and Caribbean



Federal fishing permit required in federal waters. Purchase at hmspermits.noaa.gov.

HMS recreational permit holders that fish for sharks will need to obtain a shark endorsement.

Authorized species	Minimum size (fork lenth)	Bag limit (per trip)
Smoothhound Shark	None	None
Atlantic sharpnose Shark	None	1 per person
Bonnethead Shark	None	1 per person
Shortfin Mako Shark	71 inches male 83 inches female	1 shortfin mako, hammerhead, or other shark per vessel
Hammerheads (great, scalloped, and smooth)	78 inches	
Other sharks	54 inches	

Recreational anglers are required to use non-offset, non-stainless steel circle hooks except when fishing with flies or artificial lures.

All ridgeback sharks are prohibited, except **Tiger, Oceanic Whitetip, and Smoothhound.**

Prohibited ridgeback sharks include **Bignose, Caribbean Reef, Dusky, Galapagos, Night, Sandbar,** and **Silky**. For more details on prohibited species, please refer to the Prohibited Species Placard.

Ridgeback sharks have an **interdorsal ridge** (a visible line, or crease of raised skin between dorsal fins)

Tiger Shark: Snout length much shorter than mouth width; markings fade with age; max. size 15 feet; coastal and offshore

Oceanic Whitetip Shark: Young sharks have black mottling on most fins; does not always have interdorsal ridge; max. size 8 ft; offshore

Cannot be retained if tuna, swordfish, or billfish are onboard

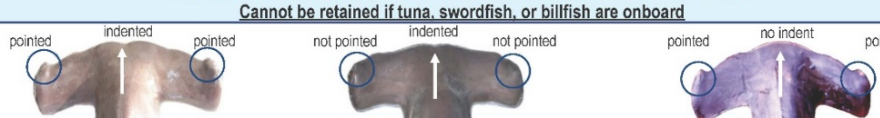
Smoothhound: (a.k.a. smooth dogfish and Florida/Gulf smoothhound) Pre-dorsal ridge present; second dorsal fin slightly smaller than first dorsal fin and much larger than anal fin; max. size 5ft; coastal and offshore

Spiny Dogfish: Max. size 4 ft; coastal and offshore

There are no recreational restrictions for **Spiny Dogfish**

Scalloped Hammerheads, Great Hammerheads, and Smooth Hammerheads

Cannot be retained if tuna, swordfish, or billfish are onboard

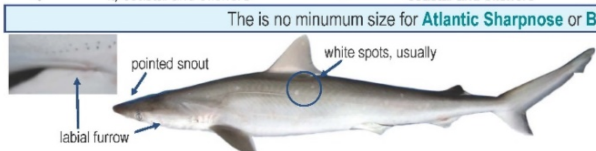


Scalloped Hammerhead: Max. size 11 ft; coastal and offshore

Great Hammerhead: Max. size 15 ft; coastal and offshore

Smooth Hammerhead: Max. size 12 ft; coastal and offshore

The is no minumum size for **Atlantic Sharpnose** or **Bonnethead Sharks**



Atlantic Sharpnose Shark: Max: size 3 ft; coastal and offshore; similar species: **Smalltail** sharks have very reduced labial furrows and **Caribbean Sharpnose** sharks lack white spots.

Bonnethead: small, black spots on body; max. size 4 ft; mostly coastal

All sharks are not identical. These are common characteristics. Young sharks can vary in appearance from adults. Maximum sizes are approximate.

Photographs and illustrations provided by NMFS, J. Castro, W.B. Driggers III, E.R. Hoffmayer, and S. Iglésias. Prohibited species are underlined in red

https://www.fisheries.noaa.gov/topic/atlantic-highly-migratory-species. Revised March 2019

If you don't know, let it go

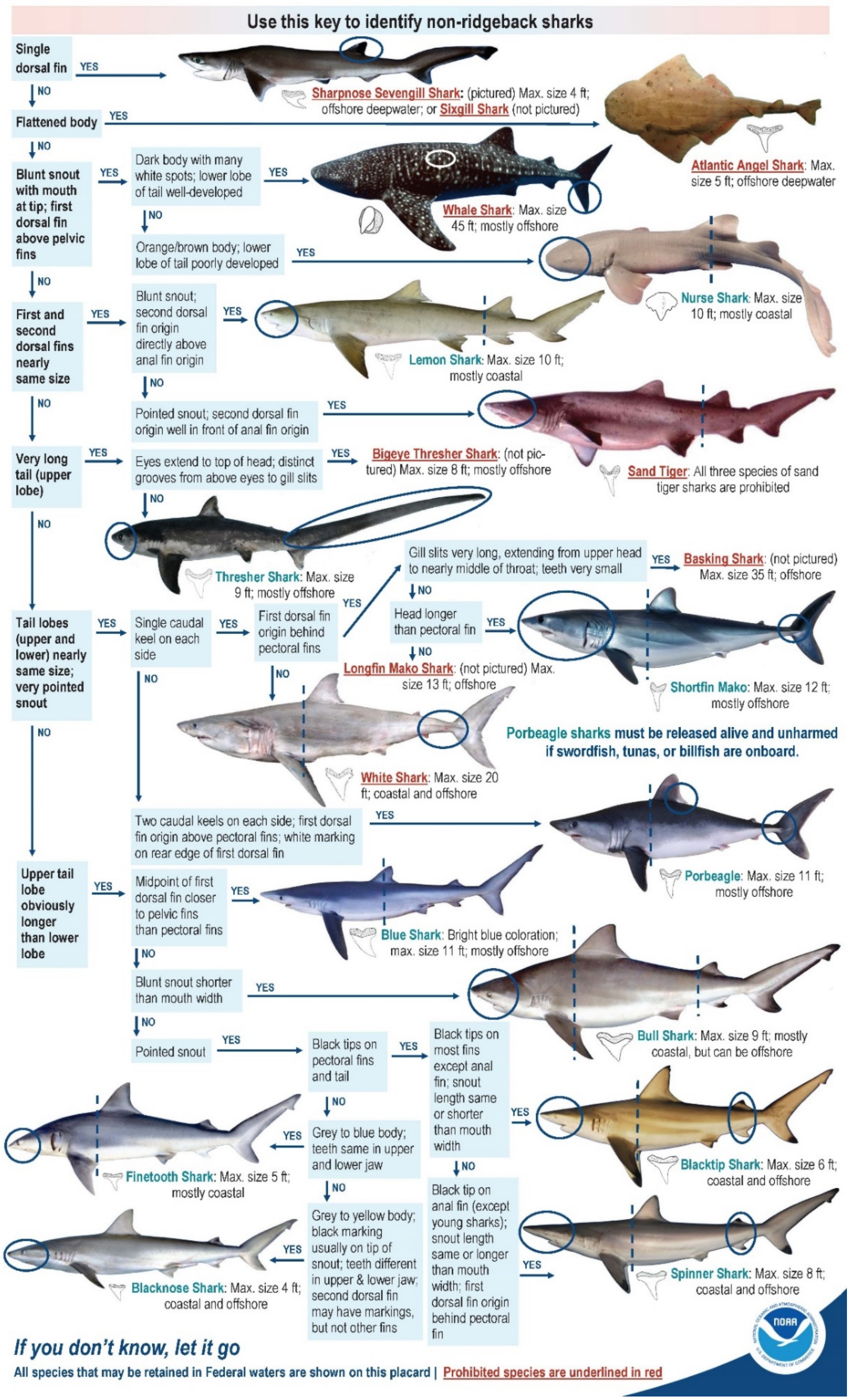


Figure 82. Shark Identification and Federal Regulations for the Recreational Fishery of the U.S. Atlantic, Gulf of Mexico, and Caribbean
(https://www.fisheries.noaa.gov/resource/outreach-and-education/shark-identification-placard)

Appendix I. Relocation Trawling Net Guidance

The relocation net specifications for a lazy line required in Appendix B Section 3.5 are based on the guidance in the memo included in this appendix.

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southeast Fisheries Science Center
Mississippi Laboratories
P.O. Drawer 1207
Pascagoula, MS 39568-1207

November 30, 2007

MEMORANDUM FOR: Stacy Carlson, Protected Resources Divison, SERO

FROM: John Mitchell & Wendy Taylor
Harvesting Systems Branch, SEFSC

SUBJECT: Recommendations for lazyline configuration to reduce dolphin interactions with turtle relocation trawlers

This memo is in reference to the conference call of November 5th during which representatives from SERO Protected Species Branch, SEFSC Harvesting Systems Branch, Army COE and several sea turtle relocation contract companies discussed methods of preventing the accidental entanglement of dolphins in the lazylines of contracted sea turtle relocation trawlers. At the conclusion of the call, I offered to summarize the various techniques which were discussed in order to provide a reference for contractors who may wish to trial a particular method during future turtle relocation work. NOAA Fisheries would like to acknowledge the efforts of Captain Steve Bosarge, of Bosarge Boats, Pascagoula Mississippi, for allowing us to provide details of his method of lazyline rigging which seems to have real potential for reducing dolphin entanglement.

Conventional lazyline rigging (Figure 1)

Conventional lazylines are attached at their forward end to the top/back edge of the inside trawl door and at their aft end to a ring in the "elephant ear", a triangle of reinforced webbing sewn to the trawl bag which acts as a lifting strap (Figure 1). The length of the lazyline is dependent on trawl size. As an example, a 55-ft. headrope length trawl might have a lazyline which is 80 to 85 feet in length. The conventional lazyline must be of sufficient length so as to allow the line to be hauled to the side of the boat upon haul back. The line is then led through a snatch block and wound around a cat head to in order to lift the bag to the side of the boat and eventually emptied on deck. When in a fishing configuration, the ample length of the lazyline forms a 10-12 foot loop behind the tailbag. This loop floats even with, or slightly above and behind the tailbag. It is in this loop of the lazyline, near the trawl bag, that underwater video obtained by NOAA Harvesting Systems gear researchers has documented dolphin interactions while the trawl is fishing. The animals appear to be using the line as a scratching post, moving back and forth against the line, rubbing it along their backs and bellies. Based on our observations, it is conceivable that a dolphin could put a complete wrap of the lazyline around it's torso. With the tail flukes acting as a stop, the animal would be unable to free itself. The following alternate lazyline rigging methods offer potential means to prevent dolphin entanglement.

Method A : Conventional rigging using stiffer lazyline (PolyDAC or Polyester)

This method simply replaces the conventional ¾" to 1" diameter polypropylene rope typically used for shrimp trawl lazylines with line made from polyDAC (a combination of polypropylene and polyester) or a polyester line. Both lines should be made with a "crab lay" which is a term relating to stiffness of the rope. Crab lay rope is used in the commercial crab fishery of the Alaska and provides an exceptionally tight twist to the rope to prevent loops when coiling. Both a polyDAC and polyester ¾" crab lay line were evaluated by Harvesting Systems divers in June, 2007 during annual gear tests. Divers found the polyDAC and polyester lines to be significantly stiffer and less pliable underwater than the conventional polypropylene lines. It was difficult for divers to form loops in either of the two line types. Divers also noted that the polypropylene arced upward (positive buoyancy) and polydacron arced downward (negative buoyancy) while being towed.

PolyDAC or Polyester ¾" crab lay line can be purchased from major U.S. rope suppliers such as TrawlWorks, Sampson and others. A recent check of cost resulted in a price of between 50 to 60 cents/foot for either rope type.

Method B: Bosarge Method of lazyline configuration (Figure 2).

This method replaces the conventional polypropylene lazyline with a stainless steel 3/8" cable. Steve Bosarge provides a very good description of the method in the following text:

"The modification to the original system totally removes the standard lazy lines and sugar line and replaces them with two 3/8 inch by 250 foot stainless steel cables in an entirely different routing arrangement. Now the routing begins with the stainless steel cables spooled on a deck winch. From there the cables run to the top of the boom where they go through separate blocks that are affixed to the boom. The cables then run to the webbing strap (elephant ear). This new system has benefited both the crew and the observers aboard the vessel by making gear retrieval faster and safer in inclement weather."

Steve's method of rigging the lazyline is likely to be much safer for dolphins for the following reasons; 1.) The diameter of the cable is significantly smaller than standard polypropylene lazyline, and thus is probably less enticing to dolphins for rubbing and scratching interactions; 2.) The fishing angle of the cable is likely higher than that of the conventional lazyline due to the cable being led from the vessel boom. We think it would be difficult for dolphins to interact with the cable at a higher fishing angle because they would have to orient themselves somewhat vertically while maintaining forward motion to do so.3.) In comparison to conventional polypropylene line, cable requires more force/exertion in order to form a tight loop when fishing, thus decreasing the probability of entanglement.

As Steve mentions above, operational benefits to the Bosarge method include quicker and safer retrieval of the tailbags (time is not lost to crewmembers having to grapple the lazylines, feed the lines through blocks and then wind them around winch cat heads to bring bags aboard). With the cable method, both tailbags are winched aboard and positioned over the deck for emptying in one simultaneous and continuous operation.

For the average shrimp fisher, there is an additional expense and some amount of special rigging which is required to use the cable method. First, a dedicated winch, capable of hauling in two tailbags simultaneously is needed. It is very likely that a vessels' try-net winch could be used for this purpose. Approximately 500 ft. of 3/8" stainless steel cable is required (estimated cost $750.00) as well as two (2), 5-ton "fat-boy" style try-net blocks (estimated cost $100.00 ea.).

Method C: Bosarge Method of lazyline configuration using PolyDAC or PE "crab lay" line

This method is identical in all respects to the above (Method B), with the replacement of 3/8" stainless steel cable with PolyDAC or Polyethylene "crab lay" line. Routing of the lazyline is the same as in the cable method. For the contractor, the advantage to this method over Method B is that a dedicated winch for the lazyline retrieval is not necessary. The trawl winch cat heads could be used to retrieve the lazyline. When fishing, the lazylines could be "tied-off" to the pin rail just aft of the trawl winches. Because the lines are routed through the boom, as in the cable method, we expect that the in-water fishing angle of the line would be higher (like the cable). The higher fishing angle along with stiffer lazyline should lessen the potential for dolphin entanglement over a standard lazyline configuration.

In FY08, the Harvesting Systems Branch plans to conduct qualitative assessments of all three of the above methods during TED evaluations aboard the *R/V Georgia Bulldog* (Feb.-April) using trawl-mounted underwater video cameras, and a DIDSON scanning sonar system. Additionally, we plan to conduct diver assessments and obtain underwater video of all three methods during annual gear evaluations using NOAA SCUBA divers in Panama City, Florida (June, 2008).

We would greatly appreciate feedback from the turtle relocation contractors who may trial the above mitigation methods. We are especially interested to know the following.

- Do contractors feel the methods are effective at preventing dolphin interactions/entanglements?
- What are the advantages and or disadvantages to working with PolyDAC / PE crab lay line during normal relocation operations?
- Is the stiffness of the the PolyDAC / PE crab lay line resilient, or is it degraded over the course of normal operations, i.e. winding around cat heads?
- Are their additional methods and or rigging modifications that may be worth investigating?

Responses can be emailed to my attention at john.mitchell@noaa.gov.

We look forward to continued collaboration with you, and the relocation contractors on this issue and hope that you will not hesitate to contact us if you have any questions.

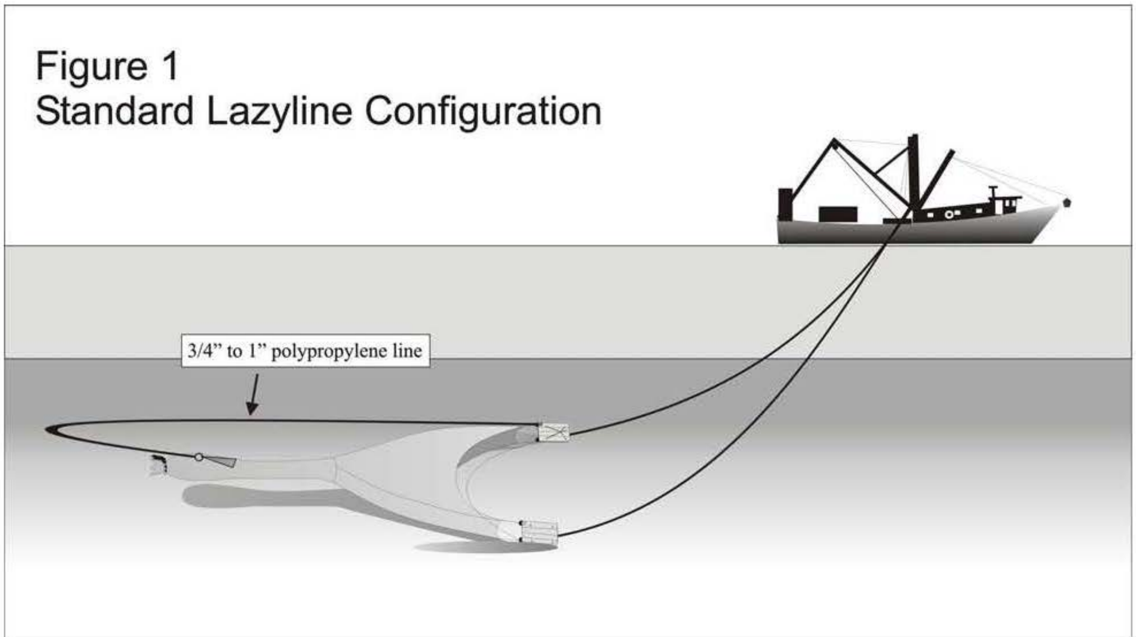


Figure 1
Standard Lazyline Configuration

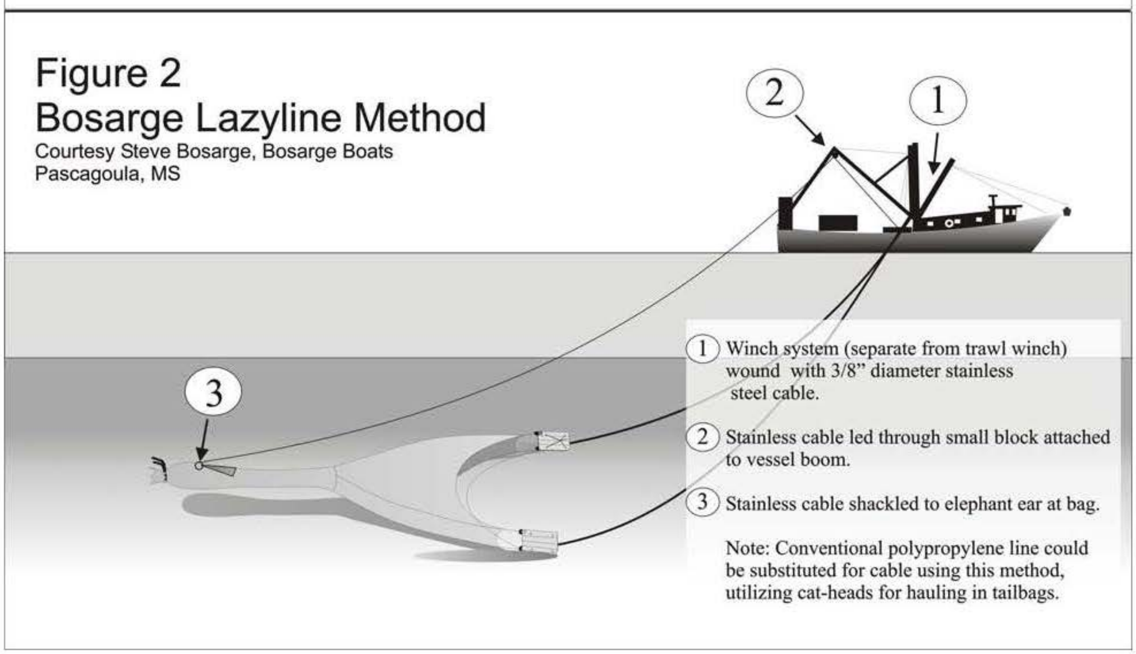


Figure 2
Bosarge Lazyline Method
Courtesy Steve Bosarge, Bosarge Boats
Pascagoula, MS

Appendix J. NMFS Recommendations to Consider when Developing the Risk Assessment for the 2020 SARBO

The USACE and/or BOEM will develop a risk assessment plan as outlined in the 2020 SARBO Section 2.9 that will incorporate information provided by the SARBO Team, including information on species presence in different areas and at different times, among other items. This appendix provides NMFS suggestions of specific factors for USACE and BOEM to consider with initial project planning, including items already discussed with the SARBO Team that are likely to be relevant to the risk assessment. A general description on the ways in which the USACE and/or BOEM will use the risk-based assessment are outlined in the 2020 SARBO Section 2.92020.

1 Pre-Construction Risk Assessment

1.1 Example Factors Considered in Pre-Construction Risk Assessment

The USACE and BOEM consider many factors when determining when and how a project should be completed. Considerations may include the urgency to complete work (e.g. current or expected traffic restrictions), dredge quantity expected, or the length of time needed to complete the project.

A. Assess the potential risk to ESA-listed species by evaluating available information on species trends and historic use of the area:
   - Review the timing and location of aggregation areas, migration through the project area, spawning or nesting seasons.
     - Sea turtles may be at higher densities during spring and fall migration along the coast.
     - Green sea turtles may be at higher densities in entrance channels with groins or jetties that support macroalgae for foraging.
     - Atlantic sturgeon maybe at higher densities staging at the mouth of sturgeon rivers prior to a spawning run. Times vary by river and seasonal water temperatures.
   - Review species takes that has occurred at the project location during past dredging or placement projects. This information can be used to assess the potential risk of future encounters.
   - Review species population assessments and recovery plans which can provide additional species information and use of an area.

B. Assess the need for relocation trawling for each dredging project:
   - Review the location and timing of the project to determine the relocation trawling effort needed and if trawling should begin prior to or after construction has commenced.
   - Review species take that has occurred at the project location during past relocation trawling to determine the potential density and relocation trawling effort needed.

C. Assess any timing considerations
   - Sea turtles: To minimize the risk of hopper dredging take of ESA-listed sea turtles, water temperature should be considered. Completing projects, such as hopper dredging, when

water temperatures are colder and sea turtles are less abundant, may reduce the risk of take. However, when water temperature drops below 11º C, sea turtles are at risk of being cold-stunned, which increases the risk of take by any equipment type (see cutterhead analysis in the 2020 SARBO Section 3.1.1.4.1). Sea turtle temporal and spatial behavior patterns should be considered when evaluating trawling, and/or dredging activities. A higher abundance of sea turtles may occur in certain locations during the Spring (March and April) and Fall (September and October) time periods as they migrate along the coast.

- North Atlantic right whales: Large dredging projects that require the use of high-speed survey vessels (similar to the Florida II) should avoid working during winter months when North Atlantic right whales are present, as outlined in the North Atlantic Right Whale Conservation Plan (Appendix F), to the extent practicable.
- Sea turtle nesting beaches: While the USACE is not proposing placement on beaches during sea turtle nesting season at this time, beach placement requires coordination with the U.S. Fish and Wildlife Service, as NMFS jurisdiction over listed sea turtles is limited to the marine environment.
- Sturgeon rivers: Project timing must consider the sturgeon specific PDCs and the information available for sturgeon rivers in Table 56 in Appendix E to minimize the risk of take.

D. Assess the equipment options available to reduce take
   - Review the level of past take in the general area based on the equipment type used and the site conditions.
   - Can the clean-up phase be completed by bed-leveling to reduce take?
   - Would cutterhead dredging reduce turbidity in the area and minimize the risk of sedimentation issues for nearby coral and seagrasses?

E. Review take: including the species taken, number of takes for the entire project, the rate of takes (e.g., multiple in 1 day, daily, weekly), and number of takes for that species compared to the take limits allowed under SARBO.

F. Review any operational issues:
   - Review the equipment to ensure it is working properly and that the drag head or cutterhead are remaining embedded in the sediment to the maximum extent practicable, as defined in the PDCs of the 2020 SARBO. This may include reviewing the USACE Dredging Quality Management data collection system used to monitor dredging operations and dredged material placement. USACE Dredging Quality Management data may be used to generate graphs as part of the risk assessments to determine if the dragheads are operating properly and only pumping when imbedded in the sediment.
   - Evaluate the channel or borrow area conditions to assess whether the physical conditions of the dredging environment are impacting dredging operations and increasing the risk of take.

G. Consider the progress of the project, such as how close is it to completion, to determine if the risk of continuing is appropriate.

H. Condition of channel or borrow areas: Determine if the work can be deferred to another time or if the dredge can work in an alternate position of the channel or borrow area that promotes more efficient dredging operations and reduces entrainment risk.

I. Clean up: If the take occurred during the cleanup phase, consider if bed-leveling or other cleanup dredging options are available that have less risk of take.

J. Relocation trawling:
   - Determine if relocation trawling should begin, if it is not already occurring.
   - If trawling is already occurring, does the effort need to be increased?

Appendix K. SARBO Additional Consultation History

Due the long and complex history of dredging and material placement within the action area, the consultation history is provided here as an appendix. It includes many types of dredging equipment and restrictions are included in the history that are explained in greater detail in the 2020 SARBO Section 2. The USACE Engineering Manual (EM) 1110-2-5025, Dredging and Dredged Material Management, (USACE 2015b) also provides more detail on the USACE navigation dredging program, dredge equipment, and dredge material management. The consultation history was provided in most part by the USACE and BOEM as a part of the SARBO Biological Assessment (SARBA) provided in 2008 and updated in 2017.

1 USACE Consultation History

1.1 Prior to 1991

USACE has been responsible for the development and maintenance of navigable waterways in the U.S. since the 1800's. The role of USACE with respect to navigation is to provide safe, reliable, and efficient waterborne transportation systems for the movement of commerce, national security needs, and recreation (Verna and Pointon 2000).

Prior to 1991, each USACE SAD District (i.e., Wilmington District [SAW], Charleston District [SAC], Savannah District [SAS], and Jacksonville District [SAJ]) prepared individual project Biological Assessments and consulted with NMFS under ESA Section 7 for each dredging event. NMFS later determined that individual consultations did not accurately assess the regional implications of dredging actions and suggested that a single regional Biological Opinion to address the dredging of channels and subsequent beach nourishment activities along the Atlantic coast from North Carolina through Florida was more appropriate, which the USACE agreed to and led to the development of the 1991 SARBO discussed below.

1.2 1991 SARBO

On November 25, 1991, NMFS issued a Biological Opinion on *Dredging of Channels in the Southeastern United States from North Carolina through Cape Canaveral Florida* (NMFS 1991, referred to as the 1991 SARBO) (NMFS 1991). The 1991 SARBO stated that continued unrestricted hopper dredging along the Southeast Region's Atlantic coast could jeopardize the continued existence of listed sea turtles. Therefore, a Reasonable and Prudent Alternative provided in the Opinion included seasonal restrictions of hopper dredging from December 1st through March 31st in channels from North Carolina through Canaveral Harbor, Florida; however, seasonal restrictions could be adjusted on a channel specific basis with appropriate supporting evidence. The implementation of seasonal restrictions on hopper dredge operations proved effective in reducing sea turtle takes throughout the South Atlantic; however, more research assessing sea turtle abundance within channels was recommended in the Opinion. The 1991 SARBO concluded that hopper dredging was unlikely to adversely affect North Atlantic right whales, contingent upon the institution of precautionary measures, including a North Atlantic right whale "watch." The 1991 SARBO also concluded that other types of dredges, including clamshell, pipeline, split-hull hopper dredge Currituck, and sidecast dredges, were unlikely to adversely affect sea turtles. An ITS of 2 Kemp's ridley, or 5 green, hawksbill or leatherback turtle's mortalities, or 50 loggerhead turtle mortalities was provided. Re-initiation of consultation was triggered for any project in which 5 turtles were taken.

1.3 1995 SARBO

On August 25, 1995, NMFS issued an update to the 1991 Opinion on the *Dredging of Channels in the Southeastern United States from North Carolina through Cape Canaveral Florida* (NMFS 1995, referred to as the 1995 SARBO) (NMFS 1995), which included beach nourishment activities, prohibited regular maintenance dredging by hopper dredge in Canaveral Harbor, Florida, eliminated a dredging window from North Carolina through Pawleys Island, South Carolina, adjusted dredging windows for other harbors in the Southeast, and included an annual incidental take of 7 Kemp's Ridley sea turtles, 7 green turtles, 2 hawksbill sea turtles, 20 loggerhead sea turtles, and 5 shortnose sturgeon.

The 1995 SARBO was developed in response to a new Regional Biological Assessment provided by the USACE, requesting that NMFS consider expansion of the dredging window based on: (1) the USACE conservative take record since 1991, (2) the willingness to continue dredging in the cooler months, during periods of reduced risk to sea turtles, to the maximum extent practicable, (3) the willingness to shut down when take numbers exceed anticipated numbers, and (4) the implementation of the turtle deflecting draghead on all hopper dredges to reduce take. The 1995 Opinion incorporated a 1995 study, which evaluated sea turtle abundance in 6 South Atlantic U.S. channels (Canaveral Harbor, Florida; Kings Bay, Florida; Brunswick Harbor, Georgia; Savannah Harbor, Georgia; Charleston Harbor, South Carolina; and Morehead City Harbor, North Carolina) and looked at species composition, population structure, and spatial and temporal (seasonal) distributions (Dickerson et al. 1995). This study was the first to scientifically evaluate sea turtle abundance and temporal distribution and concluded that fewer sea turtles were captured when water temperatures were at or below 16°C; thus, 16°C was recommended as a conservative threshold indicator for a reduced risk of sea turtle take during hopper dredging operations in the Atlantic. Though this study helped define water temperature as a critical factor in sea turtle occurrence within these 6 channels, additional studies were recommended to refine site specific factors that may influence sea turtle presence (i.e. immigration/emigration periods, influence of coastal dynamics on water temperature, etc.).

Recognizing the new information gathered by Dickerson et al. (1995) and subsequent documentation in the USACE Regional Biological Assessment, the 1995 Opinion (NMFS 1995) did not include seasonal hopper dredging restrictions from Pawley's Island, South Carolina, through North Carolina or Titusville, Florida to Key West, Florida. Furthermore, the dredging windows from Pawley's Island, South Carolina to Tybee Island, Georgia and from Tybee Island, Georgia to Titusville, Florida were expanded.

However, a series of new Terms and Conditions were identified in the 1995 SARBO to minimize take including an emphasis on the incorporation of the new turtle deflecting draghead on hopper dredges as a mechanism to reduce takes.

1.4 1997 SARBO

On April 9, 1997, NMFS issued an Interim Opinion on *Continued Hopper Dredging of Two Channels and Two Borrow Areas in the Southeastern United States During 1997* (NMFS 1997). The USACE hopper dredging activities in channels and borrow areas along the southeastern coast of the United States during the spring of 1997 resulted in an unanticipated high rate of sea turtle takes and were rapidly approaching the incidental take level established in the 1995 SARBO, which required re-initiation of consultation if more than 1 turtle was taken in any day, or 5 were taken in any single channel. Re-initiation of formal consultation was required because 75% of the incidental take limit had been reached. The interim consultation addressed the use of hopper dredges during 1997 in the Atlantic portion of the USACE SAD, including Morehead City Harbor, Wilmington Harbor, and the Myrtle Beach and Miami borrow areas (NMFS 1997). For these 3 projects, the annual incidental take total for loggerhead sea turtles was increased by 15.

On September 25, 1997, NMFS issued an Opinion on the *Continued Hopper Dredging of Channels and Borrow Areas in the Southeastern United States* (NMFS 1997), referred to as the 1997 SARBO. This Opinion was written to amend the 1995 SARBO and supersede the 1997 Interim Opinion (NMFS 1997). It set an annual documented incidental take of 7 Kemp's ridley sea turtles, 7 green sea turtles, 2 hawksbill sea turtles, 35 loggerhead seat turtles, and 5 shortnose sturgeon, and clarified monitoring requirements for beach nourishment projects. Furthermore, the 1997 SARBO required hopper dredge windows (established in the 1995 SARBO), along with the requirement to minimize sea turtle takes by refining the turtle deflecting dragheads on hopper dredges, continued working in the cool water months to the maximum extent practicable, and agreed to shut down operations when high numbers of turtle takes occurred before approaching the incidental take limit for a given species (NMFS 1997). Prior to the release of the Opinion, USACE implemented a USACE SAD protocol to effectively manage take numbers throughout each USACE District. This protocol expressed the USACE commitment to dredge during cool water periods in channels where sea turtle abundance is high or where substrates render the turtle deflecting draghead ineffective.

2 Reinitiation of Consultation for the 2020 SARBO

2.1 2006

On March 23, 2006, the USACE Jacksonville District, submitted a request to initiate consultation with NMFS under Section 7 of the ESA for the use of bed-leveling devices for associated cleanup activities after cutterhead, hopper, and mechanical dredging in 4 ports on the east coast of Florida. In a letter dated June 23, 2006, NMFS made the determination that bed-leveling is not part of the hopper dredging activity previously consulted on in the 1997 SARBO and as such, re-initiation of consultation was required pursuant to 50 CFR§ 402.16(c) ("if the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the Biological Opinion."). Furthermore, NMFS determined that reinitiation was also appropriate under 50 CFR § 402.16(d), because "a new species is listed or critical habitat designated that may be affected by the identified action."

On July 6, 2006, NMFS sent an e-mail to the SAJ providing guidance on the procedures the USACE should follow during re-initiation in order to cover the current use of bed-levelers. On December 29, 2006, NMFS sent a letter to USACE SAJ notifying the SAJ of NMFS's intent to phase out the use of Section 10(a)(1)(A) permits for relocation of sea turtles near hopper dredging activities, by letting current permits expire. NMFS recommended the USACE reinitiate consultation for dredging activities in the south Atlantic such that a new Opinion and ITS would be a more appropriate mechanism to cover relocation trawling take associated with dredging activities. Informal communication and coordination with NMFS began with the USACE at this time and on April 30, 2007, the USACE SAD sent a letter to NMFS formally requesting re-initiation of consultation under Section 7 of the Endangered Species Act (ESA), in regard to the 1997 SARBO (NMFS 1997).

2.2 2008

On September 12, 2008, USACE SAD and BOEM provided a jointly prepared South Atlantic Regional Biological Assessment (USACE and USEPA 2008) to provide new information and analysis for a revised and updated SARBO. NMFS received the letter on September 15, 2008 and considers this date the official request for consultation.

2.3 BOEM Consultation History

Pursuant to Public Law 103-426 (43 U.S.C. 1337(k)(2)) (1994), BOEM can authorize the use of Outer Continental Shelf (OCS) sand, gravel, or shell resources in approved projects. Potential borrow areas are typically identified during the feasibility phase of beach nourishment, coastal restoration, or construction projects, a process which generally occurs well in advance of formal application to the BOEM for the use of OCS sand, gravel, and shell resources.

BOEM involvement with ESA Section 7 consultations with NMFS regarding offshore dredging activities in the South Atlantic dates to 1995 with the Duval County Shore Protection Project (Florida) and Myrtle Beach Storm Risk management Project (South Carolina). Since that time, BOEM has continued to consult with NMFS on an individual project basis for dredging activities in the OCS while awaiting inclusion of those activities in 2020 SARBO.

2.4 USACE and BOEM Consultation History 2008-2016

Since the submission of the 2008 SARBA, USACE and BOEM staff have continued to work with NMFS staff to respond to data needs, newly listed species, and information requests. The SARBO Team (consisting of members of NMFS, USACE, and BOEM staff) was formed to work together to determine the scope and scale of activities that should be included in the 2020 SARBO and how to improve the dredging program covered under SARBO while minimizing impacts to ESA-listed species and designated critical habitat. The list below provides a highlight of the interim steps and accomplishments to complete this task.

- July 1, 2010: NMFS submitted a request to USACE SAD for additional information and raised certain areas of concern. In a letter dated August 9, 2010, USACE SAD provided NMFS with the requested information, acknowledging unresolved issues.

- July 2012: USACE SAD and NMFS staff met to discuss revising and updating information for the 2020 SARBO.
- November 6, 2013: NMFS sent a second request for information letter resulting in further coordination NMFS to clarify the request and provide appropriate information.
- February 7, 2014: USACE SAD submitted a request to NMFS asking for NMFS to issue an "interim supplement" to the 1997 SARBO to address recent listings of the Atlantic sturgeon (New York Bight, Chesapeake Bay, Carolina, and South Atlantic, and Gulf of Maine DPS) and smalltooth sawfish (U.S. DPS) and to provide incidental take for Atlantic sturgeon (all 5 DPSs). NMFS declined the request for an "interim supplement," but clarified that while the reinitiation of consultation is ongoing, SAD should continue to conduct dredging operations under the Terms and Conditions of the 1997 SARBO and SAD's internal protocols for managing endangered species interactions with dredging. NMFS Southeast Region also recommended the preparation of an Endangered Species Act Section 7(a)(2)/7(d) jeopardy analysis. In response to this guidance, the USACE and BOEM have provided 7(a)(2)/7(d) analysis for all species and designated critical habitat listed since the completion of the 1997 SARBO.
- June 27, 2017: USACE SAD and BOEM provided NMFS an updated biological assessment (USACE 2017), including proposed Project Design Criteria (PDCs) necessary for the competition of a Programmatic Biological Opinion. As discussed between NMFS, USACE, and BOEM, the goal of framing this Biological Opinion as a Programmatic Opinion is to expand the scope of the dredging and dredge material placement activities considered, provide limits of how dredging and dredge material placement activities will occur through the use of PDCs, and to allow some flexibility in how these activities are completed in the future. The updated SARBA incorporated all 7(a)(2)/7(d) analysis for all species and designated critical habitat listed since the completion of the 1997 SARBO.
- August 6, 2017: USACE SAD, BOEM, and NMFS met in Atlanta, Georgia, to work toward resolution on outstanding issues and complete the required PDCs.
- February 2, 2018: USACE provided a quantitative analysis of anticipated impacts to ESA-listed species and critical habitat.
- April 25, 2018: USACE SAD sent a copy of GIS layers to NMFS for internal coordination needed to complete the 2020 SARBO that showed areas where activities are expected to continue to be dredged or have material placement.
- July 17-18, 2018: USACE SAD, BOEM, and NMFS senior leadership met in St. Petersburg, Florida to work toward resolution on outstanding issues and completed the required PDCs, specifically negotiating protection for North Atlantic right whale, Atlantic sturgeon, and ESA-listed corals. A completion date for 2020 SARBO was set for the summer of 2019.
- August 28, 2018: USACE SAD, BOEM, and NMFS met in Atlanta, Georgia, with state partners to solicit feedback on the draft PDCs and request any additional regional information available to improve protections of ESA-listed species.
- April 19, 2019: USACE SAD, BOEM, and NMFS met in St. Petersburg Florida to finalize negotiations for outstanding issues that included resolution of the necessary protections for

ESA-listed coral, Atlantic and shortnose sturgeon, Johnson's seagrass, and to North Atlantic right whales from geophysical surveys.

- September 13, 2019, the USACE SAD, BOEM, and NMFS finalized the mutually agreed upon PDCs for this Opinion and initiated consultation.[98]
- October 25, 2019, a draft of the Biological Opinion (NMFS tracking number SERO-2008-00000) was sent to the USACE and BOEM be provided to the USACE and BOEM prior to final signature.
- November 1, 2019, USACE provided comments to NMFS on the draft Opinion. Requested changes to the type of equipment allowed in sturgeon rivers and the timing of work covered under this Opinion. Also requested expanding the frequency of geophysical survey equipment covered under this Opinion. USACE stated that the ITS did not seem to accurately reflect the level of work that will be covered under this Opinion and requested that it be recalculated. USACE requested to review another draft to see the edits made to their substantial comments prior to finalization of the Opinion.
- November 8, 2019, BOEM provided comments to NMFS on the draft Opinion.

2.5 Key Information Gathered and Decisions Made by the SARBO Team

Following completion of the 2008 SARBA, the SARBO Team worked together to further define and expand the types of projects included under the 2020 SARBO and to collaboratively develop the PDCs necessary to be protective of ESA-listed species and designated critical habitats. The updated SARBA provided by the USACE and BOEM to NMFS on June 27, 2017, described changes proposed at that time and documented the first round of protective measures developed. These measures were revised through numerous calls, in-person meetings, and shared drafts of the PDCs and proposed action. Key information gathered and considered during this further development of the proposed action, including those measures protective of ESA-listed species and designated critical habitats, is summarized below. The details of the proposed action are discussed further in the 2020 SARBO Section 2.

2.5.1 Expanding Where Projects Could Occur from 1997 SARBO

The 2020 SARBO action area extends beyond the action area considered in previous iterations of SARBO, in part due to inclusion of projects in federal waters under the jurisdiction of BOEM. The action area for 2020 SARBO specifically includes the following areas and projects that were not considered in the 1997 SARBO:

- Federal and territorial waters in the U.S. Caribbean (Puerto Rico and the U.S. Virgin Islands).
- Projects within critical habitat designated after completion of the 1997 SARBO (*Acropor*a critical habitat, loggerhead critical habitat, Atlantic sturgeon critical habitat, revised North Atlantic right whale critical habitat, and Johnson's seagrass critical habitat). The number of

[98] Under the ESA consultation is initiated once NMFS has all information necessary to conclude the consultation. Because the PDCs that are the core of the project description were under development through the entire process, consultation was initiated very close to the issuance of the Opinion.

projects covered under the 2020 SARBO is expected to increase from the 1997 SARBO in areas where critical habitat was designated after completion of the 1997 SARBO, including *Acropor*a critical habitat, loggerhead critical habitat, Atlantic sturgeon critical habitat, revised North Atlantic right whale critical habitat, and Johnson's seagrass critical habitat. The number of projects completed under the 1997 SARBO in Johnson's seagrass, Atlantic sturgeon, and *Acropora* critical habitat were limited to those completed under a memo in which the USACE determined that completion of the project would not result in adverse modification to critical habitat, under ESA Section 7(a)(2)/7(d). Development of PDCs to protect these areas of designated critical habitat required extensive coordination between the SARBO Team and information gathering from other entities, including the species experts and state representatives to provide consistency with each state's regulations where possible.

- Maintenance dredging permitted by the USACE in areas not previously considered under the 1997 SARBO. The 2020 SARBO covers maintenance dredging actions in any area within the action area (2020 SARBO Section 2.8) as long as all relevant 2020 SARBO PDCs are met. This Opinion provides the definition of maintenance dredging and specific limitations in the SARBO PDCs. Generally, the 2020 SARBO will expand the areas that can be maintenance dredged, when funded, authorized, or carried out by USACE or BOEM. to include:
    - Navigational waterways required to be maintained under Title 33 (Navigation and Navigable Waters) that were initially included in 1997 SARBO, but were located in areas where subsequently listed species or designated critical habitat occur: Maintenance of these major navigation channels maintained as navigational waterways such as the Intracoastal Waterway (IWW) were covered under the 1997 SARBO, but effects to species listed and critical habitat designated following the completion of 1997 SARBO were not. These channels will be covered under the 2020 SARBO. For example, certain portions of the IWW occur in areas where critical habitat was designated after the completion of the 1997 SARBO, such as the portion of the IWW on the east coast of Florida within the range of Johnson's seagrass (listed in 1998 shortly after completion of the 1997 SARBO) and in Johnson's seagrass critical habitat designated in 2000. Maintenance of this portion of the IWW was later covered under a USACE SAJ Regulatory Division regional general permit in 2001 (NMFS 2001a)(SAJ-93 *Maintenance Dredging of the Ports and Intracoastal Waterway within the range of Johnson's Seagrass,* NMFS tracking number SER-2000-01199, signed June 4, 2001). Future maintenance dredging within the range of Johnson's seagrass in the Intracoastal waterway (IWW) will now be authorized by the USACE under the 2020 SARBO through the USACE Regulatory permitting process or be executed by the USACE Civil Works program. Other navigational waterways required to be maintained under Title 33 that are in the action area will continue to be carried out under SARBO as federally-permitted (USACE Regulatory permitted project) or as a federally-authorized (USACE Civil Works project), so long as they meet all other requirements of this Opinion.
    - Navigation channels and canals that are not required to be maintained under Title 33 are also covered such as the smaller channels or canals that provide access to coastal communities and/or coastal neighborhoods and channels in rivers that are not part of the main navigation channel such as the secondary channel sections of a braided river.

- Ports and berths along a maintained navigation channel, including those not owned and operated by a Port Authority, or federally-required to be maintained such as in the individually maintained berths in Savannah Harbor.
- Smaller areas such as public and private marinas, boat ramps, and around docks that were previously permitted by USACE and dredged.

2.5.2 Discussions Related to Project Timing

During consultation, the parties reviewed and modified seasonal work windows and restrictions based on new information that resulted in similar or improved protections for ESA-listed species This resulted in an expanded seasonal window that hopper dredging can occur under the 2020 SARBO (as compared to the 1997 SARBO), added seasonal restrictions on projects occurring in rivers where Atlantic and shortnose sturgeon occur, and modified seasonal protective measures required in areas where and when North Atlantic right whales may be present, while taking into account the need for continued protections for ESA-listed sea turtles.

Since changes in project timing can result in changes in the effects to multiple ESA-listed species and designated critical habitats, the SARBO Team coordinated with state and federal agencies to find all available data to evaluate effects. The SARBO Team met with species experts within NMFS, USACE, and BOEM, along with state partners, to request available information, discuss the effects of expanding the seasonal hopper dredging window, and to develop measures to minimize or avoid risks to ESA-listed species and designated critical habitat. Data gathered included information on (1) the risk to North Atlantic right whales from different types of equipment considered in this Opinion, and if changing the timing of projects could reduce that risk; (2) the risk to sea turtles from allowing limited dredging at times when and in areas where abundance is higher, and the risk from minimization measures like relocation trawling, and (3) the risk to Atlantic and shortnose sturgeon from modifications to project timing windows based on historic water quality data collected from rivers where sturgeon are known to be present.

Ultimately, the SARBO Team determined that the 2020 SARBO should include additional minimization measures as part of the proposed action, due to: (1) the scope and scale of this project spanning from North Carolina to the Caribbean and from inland rivers and water ways to federal water and (2) the complexity of balancing the risks to 25 ESA-listed species and 5 designated critical habitat units, and (3) the continued federal mandate for the USACE to maintain navigational waterways and beaches. Many of the ESA-listed species within the action area have overlapping ranges and habitats, and some protective measures frequently applied to projects for certain ESA-listed species conflict with protection of other listed species or critical habitats in these overlapping areas. The SARBO Team gave extensive consideration to which ESA-listed species could be affected by an activity covered under this Opinion, the probability of exposure based on project timing and anticipated species abundance in an area, and how to maximize protections for all ESA-listed species and designated critical habitat. Additional consideration was given to species' current status and ability to recover when considering the risk to multiple species in a given area. This approach resulted in the development of the risk-based adaptive management plan outlined in 2020 SARBO Section 2.9.2 while developing the PDCs provided in the South Atlantic Regional Biological Opinion (2020 SARBO) Project Design Criteria (PDCs) Overview and General PDCs.

Developing an approach to balance the needs of all 25 ESA-listed species and designated critical habitats in the action area took over a year of intensive literature review, coordination with species experts, and negotiations between NMFS, USACE, and BOEM, as highlighted in the following paragraphs. Within the action area, potential effects to North Atlantic right whale, ESA-listed corals, and *Acropora* critical habitat were identified as being at greatest risk from the proposed action, without sufficient protective measures. A discussion of the information gathered and negotiations made related to North Atlantic right whale is summarized below. Negotiations related to ESA-listed corals and *Acropora* critical habitat relate to potential changes to water quality is summarized in 2020 SARBO Section 2.5.3.

For North Atlantic right whale, the most notable consideration in this Opinion was reviewing the size, speed, purpose, location, and timing of dredging equipment and related vessels to look for ways to reduce the probability of a potential vessel collision with this species. This included the involvement of the NMFS North Atlantic right whale species coordinator, review of all relevant literature, review of automatic identification system tracking data of vessels used for or to support dredging in the action area, coordination with species experts outside of NMFS, and multiple face-to-face meetings with NMFS, USACE, and BOEM, including senior leadership participation. The negotiations resulted in an agreement to:

1. shift a portion of hopper dredging to summer months when North Atlantic right whale are not present in the action area,

2. fund up to 2 additional aerial survey teams north of the existing Florida/Georgia survey area (off the coasts of North and South Carolina) as needed to adequately cover the area to help identify when whales are actively or possibly migrating and where these whales are in the action area, in addition to the existing aerial survey team that surveys southern Georgia and northern Florida (described in Appendix F) , and

3. expand the time and distance that USACE vessels must reduce speeds when North Atlantic right whales are known to be in the area.

These increased protective measures for North Atlantic right whales are outlined in the North Atlantic Right Whale Conservation Plan (Appendix F). An added benefit to the increased aerial surveys is that when North Atlantic right whales are observed, those locations will be broadcast to other commercial vessel traffic in the area to alert mariners to be on the lookout for this species. Surveys will also aid in the reporting of other large ESA-listed species such as the newly listed giant manta ray.

The SARBO Team considered and acknowledged that shifting some dredging projects to warmer months in the negotiated North Atlantic Right Whale Conservation Plan may increase the risk of entrainment of sea turtles by hopper dredges due to the potential for higher densities of sea turtles in the project area. It was determined that the proposed action should include allowing dredging in warmer months only in limited circumstances and after a risk based assessment was completed, as outlined in the 2020 SARBO Section 2.9.2. This decision was made because even though the risk of a vessel strike to a North Atlantic right whale is low with the agreed upon PDCs, the consequence to the species from a single vessel strike would be high. Conversely, the risk of entrainment of a sea turtle increases when moving dredging from cooler winter to warmer summer months when sea turtles are more prevalent in warmer waters; however, the risk/impact to ESA-listed sea turtle species is minimized by the PDCs in this Opinion. The SARBO Team agreed to continue to work collaboratively throughout the life of the 2020 SARBO to minimize the risk of entrainment to ESA-listed sea turtles as much as possible, through adherence to negotiated protective measures outlined in the PDCs of this Opinion and through the continued risk assessment process outlined in the 2020 SARBO Section 2.9.2.

2.5.3 Discussions Related to Water Quality

The SARBO team spent years gathering information and discussing the potential effects from turbidity and sedimentation generated during dredging and material placement. Turbidity and sedimentation were of particular concern for Johnson's seagrass and ESA-listed corals. Both Johnson's seagrass and ESA-listed corals are not mobile, and therefore cannot avoid turbidity or sedimentation. As described in this Opinion, ESA-listed corals are under increased pressure range-wide from external sources resulting in the deterioration or loss of colonies by events like coral bleaching, and additional stressors, such as turbidity and sedimentation, can have high risk consequences to the status of the species. Effects to ESA-listed corals have not previously been considered in a programmatic consultation of this nature, where the exact project locations and site conditions are unknown. All of NMFS' previous Section 7 consultations involving potential effects to coral in the action area had involved reviewing the site-specific details of each project individually. Based on lessons learned from recent dredging to deepen and widen Miami Harbor (NMFS 2011a), NMFS and the USACE have been reevaluating how to estimate and quantify the extent of anticipated and observed effects to ESA-listed corals for all dredging projects. Hence, developing PDCs that accurately account for the risk of effects to ESA-listed corals at all future project locations that could be covered by this Opinion required extensive research and coordination, including involvement from the NMFS coral species team, reviewing available literature on the matter, and coordination with the local USACE SAJ and the Florida state agencies involved in state permitting processes. Specific information was gathered for this Opinion to determine the potential distance that turbidity or sedimentation can extend beyond the work area for projects considered under this Opinion and the potential areas in which corals could be relocated to avoid harm from pipelines associated with projects was used to develop the PDCs, which will be discussed and evaluated in the effects analysis of this Opinion.

2.5.4 Geophysical and Geotechnical (G&G) Surveys

The 1997 SARBO did not consider G&G surveys conducted in association with dredging projects. The USACE requested to have G&G surveys needed for dredging and material placement projects be included as part of the 2020 SARBO proposed action. BOEM completed separate consultations with NMFS on G&G surveys they conduct as part of their marine minerals program, including those activities within the 2020 SARBO action area (NMFS 2019b). Therefore, BOEM requested to be excluded from the G&G portion of this programmatic consultation, because a separate individual consultation covers BOEM's Atlantic and Gulf of Mexico OCS sand survey activities. NMFS evaluated whether BOEM's G&G survey activities meet the definition for "effects of the action", and determined that they do not. "Effects of the action" are defined as effects "caused by the proposed action, including the effects of other activities that are caused by the proposed action. An effect or activity is caused by the proposed action if it would not occur but for the proposed action and is reasonably certain to occur." (50 C.F.R. § 402.02). Thus, NMFS' determination regarding whether an effect or activity is caused by the proposed action is governed by a "but for" standard of causation. BOEM reports that while the results of G&G surveys may subsequently be used for identification of borrow areas under SARBO, the G&G surveys operate independently of the activities under consultation in SARBO, and are used for purposes beyond the 2020 SARBO proposed action. Thus, BOEM's G&G surveys within the 2020 SARBO action area do not depend on specific dredging or placement projects for justification. Because BOEM's G&G activities have utility apart from the proposed action, do not depend upon the activities included in the 2020 SARBO proposed action for justification, and would continue to occur regardless of the 2020 SARBO proposed action, we determined that BOEM's G&G activities, and related effects, are not caused by the proposed action. Therefore, NMFS did not consider BOEM's G&G surveys in this consultation. However, NMFS reviewed the Final Environmental Assessment, Sand Survey Activities for BOEM's Marine Minerals Program, Atlantic and Gulf of Mexico (BOEM 2019) and relied upon the assessment for information necessary to develop the USACE G&G PDCs and the associated effects analysis for the 2020 SARBO. NMFS also coordinated with the NMFS headquarters experts on acoustics to ensure accuracy of the G&G effects analysis from sound producing activities considered in this Opinion.

Appendix L. Revision History

July 30, 2020: Appendix G, PDC GG.4 was revised to clarify the use of single beam sonar.

On April 28, 2020 the U.S. Army Corps of Engineers (USACE) South Atlantic Division (SAD) requested that NMFS Southeast Region (SERO) revise South Atlantic Regional Biological Opinion (2020 SARBO) Appendix G Geophysical and Geotechnical (G&G) PDCs to correct an error discovered following the completion of the Opinion. Specifically, USACE requested that the limitation on operation of single beam sonar under the 2020 SARBO in PDC GG.4 be revised from requiring operation above 160 kHz to requiring operation above 24 kHz, to more accurately reflect USACE practice. The Bureau of Ocean Energy Management (BOEM) is a Co-Action Agency for SARBO and is aware of the USACE request without objection. The G&G PDCs in SARBO only apply to the USACE

The proposed revision to PDC GG.4 is noted in bold and underlined below:

G&G PDCs that apply to all geophysical surveys

GG.4 Only electromechanical sources will be used during geophysical surveys. Electromechanical sources will be limited to boomers, chirp sub-bottom profilers, side-scan sonars, and single beam, interferometric, or multibeam depth sounders.

- Survey equipment will be operated at the lowest power setting, narrowest beamwidth, and highest frequency possible to fulfill data needs and to effectively reduce exposure and received sound levels.
- Boomers and chirp sub-bottom profilers must be operated below 205 dB re 1 μPa (rms).
- **<u>Single beam depth sounders will be operated no lower than 24 kHz.</u>**
- Side-scan sonars, ~~single beam,~~ interferometric, and multibeam depth sounders will be operated above 160 kHz.
- No airguns or other deep-penetrating geophysical instruments are allowed under 2020 SARBO.

Discussion

Section 2.9.5.3 of the 2020 SARBO addresses adaptive changes to PDCs, including where a PDC is not operating or being interpreted in the manner intended. However, a PDC will only be modified if (1) there is sufficient information to support the proposed change, as determined by NOAA Office of General Counsel Southeast Region and the NMFS Southeast Regional Administrator, (2) the Regional Administrator determines that the change would not diminish the minimization of impacts of takes to ESA-listed species, and (3) the proposed change does not trigger reinitiation of this Opinion under 50 C.F.R. § 402.16.

For the reasons described below, we have determined that the proposed revision is appropriate under those criteria.

Injurious sound effects: As discussed in SARBO Section 3.1.8.4.2, single beam sonar is an intermittent non-impulsive sound sources that involves geophysical survey equipment dragged through an area, creating small, downward-focused ensonified zones. It is extremely unlikely that this highly-focused and moving sound source will be active in a particular location long enough to expose ESA-listed species to peak pressure or cumulative effects. Changes in operation frequency of single beam sonar (e.g., from 160 kHz to 24kHz) will impact only whether or not an animal can perceive a sound, but will not change the magnitude of sound experienced by ESA-listed species. We accordingly conclude that injurious effects from single beam sonar operating above 24kHz remain extremely unlikely to occur, for the reasons described in 2020 SARBO. This proposed change in PDC GG.4 does not change the analyses or outcome of the 2020 SARBO effects analysis for peak pressure or cumulative injurious sound effects.

Behavioral sound effects: The 160 kHz limitation in PDC GG.4 was selected to be out of the hearing range of all of the ESA-listed species analyzed in the 2020 SARBO, which are provided in Table 14. As discussed in SARBO Section 3.1.8.4.2, if a sound source falls outside the hearing range of a species (or harmonics are only detectable at much lower levels), it is considered inaudible and so a behavioral disturbance is considered extremely unlikely. Modifying the PDC to allow single beam sonar to operate as low as 24 kHz changes means that single beam sonar may operate within the hearing range of low-frequency cetaceans (blue, fin, and sei whales) and mid-frequency cetaceans (sperm whales). However, as discussed in SARBO Section 3.1.8.3, with the exception of North Atlantic right whales, the ESA-listed whales in the area are predominately located off of the continental shelf in waters deeper than where work covered under this Opinion would occur and would not be expected to be present in areas affected by noise from vessels and dredging covered under this Opinion. Geophysical surveys covered under the 2020 SARBO are used in association with these vessels and dredging and therefore single beam surveys are not expected to occur in areas where low-frequency and mid-frequency cetaceans occur. We therefore conclude that behavioral effects to these species would be extremely unlikely to occur. Further, in the unlikely event that these whales encounter noise associated with single beam sonar, it is expected that any exposure would be brief, given the nature of this equipment, and therefore any behavioral effects would be insignificant. SARBO PDC GG.6 limits all geophysical surveys from occurring in areas where North Atlantic right whales may be present, including single beam sonar. Single beam sonar operating at 24 kHz will still operate outside of the hearing range of other ESA-listed species in the action area. Therefore, the proposed change to PDC GG.4 does not change the outcome of the 2020 SARBO effects analysis for behavioral sound effects, as any effects remain extremely unlikely to occur.

Conclusion

For the reasons stated above, we have determined that there is sufficient information to support the proposed change to PDC GG.4. Additionally, the proposed change to PDC GG.4 does not diminish the PDCs' minimization of impacts of takes to ESA-listed species associated with 2020 SARBO, as all anticipated effects associated with PDC GG.4 as modified would remain extremely unlikely to occur. We have also determined that the proposed change to PDC GG.4 does not trigger reinitiation pursuant to 50 C.F.R. § 402.16, as the proposed modification will not result in effects to ESA-listed species or designated critical habitat beyond those considered in the 2020 SARBO.

Per the requirements of 2020 SARBO, a revised version of the Opinion with the revised PDC and supporting information, will be uploaded to the NMFS Southeast Region website and all future projects covered under SARBO 2020 will be required to adhere to the PDCs as modified.

Transmittal Letter:

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
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July 30, 2020

F/SER31:KR
SERO-2019-03111

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Dear Drs. McCallister and Reidenauer:

On April 28, 2020, we received a request from the United States Army Corps of Engineers (USACE) to revise the Project Design Criteria (PDC) for geotechnical and geophysical (G&G) surveys in the 2020 South Atlantic Regional Biological Opinion for Dredging and Material Placement Activities in the Southeast United States (2020 SARBO). Specifically PDC GG.4, in Appendix G, did not take into account single beam sonar as intended. USACE South Atlantic Division (SAD) determined following the completion of 2020 SARBO that operating single beam sonar above 160 kHz in all waters in the 2020 SARBO action area as required by PDC GG.4 would significantly reduce the actions that would be covered under the 2020 SARBO. USACE accordingly requested that NMFS revise the 2020 SARBO PDC GG.4 as follows (addition in bold/underline, strikethrough to be removed):

GG.4 Only electromechanical sources will be used during geophysical surveys. Electromechanical sources will be limited to boomers, chirp sub-bottom profilers, side-scan sonars, and single beam, interferometric, or multibeam depth sounders.

- Survey equipment will be operated at the lowest power setting, narrowest beamwidth, and highest frequency possible to fulfill data needs and to effectively reduce exposure and received sound levels.
- Boomers and chirp sub-bottom profilers must be operated below 205 dB re 1 μPa (rms).
- **<u>Single beam depth sounders will be operated no lower than 24 kHz</u>**.
- Side-scan sonars, ~~single beam,~~ interferometric, and multibeam depth sounders will be operated above 160 kHz.
- No airguns or other deep-penetrating geophysical instruments are allowed under 2020 SARBO.

NMFS has analyzed this change and determined that this revision does not change the effects resulting from activities carried out under the 2020 SARBO for the following reasons:

Section 2.9.5.3 of the 2020 SARBO addresses adaptive changes to PDCs, including where a PDC is not operating or being interpreted in the manner intended. However, a PDC will only be modified if (1) there is sufficient information to support the proposed change, as determined by NOAA Office of General Counsel Southeast Region and the NMFS Southeast Regional Administrator, (2) the Regional Administrator determines that the change would not diminish the minimization of impacts of takes to ESA-listed species, and (3) the proposed change does not trigger reinitiation of this Opinion under 50 C.F.R. § 402.16.

For the reasons described below, we have determined that the proposed revision is appropriate under those criteria.

Injurious sound effects: As discussed in SARBO Section 3.1.8.4.2, single beam sonar is an intermittent non-impulsive sound sources that involves geophysical survey equipment dragged through an area, creating small, downward-focused ensonified zones. It is extremely unlikely that this highly-focused and moving sound source will be active in a particular location long enough to expose ESA-listed species to peak pressure or cumulative effects. Changes in operation frequency of single beam sonar (e.g., from 160 kHz to 24kHz) will impact only whether or not an animal can perceive a sound, but will not change the magnitude of sound experienced by ESA-listed species. We accordingly conclude that injurious effects from single beam sonar operating above 24kHz remain extremely unlikely to occur, for the reasons described in 2020 SARBO. This proposed change in PDC GG.4 does not change the analyses or outcome of the 2020 SARBO effects analysis for peak pressure or cumulative injurious sound effects.

Behavioral sound effects: The 160 kHz limitation in PDC GG.4 was selected to be out of the hearing range of all of the ESA-listed species analyzed in the 2020 SARBO, which are provided in Table 14. As discussed in SARBO Section 3.1.8.4.2, if a sound source falls outside the hearing range of a species (or harmonics are only detectable at much lower levels), it is considered inaudible and so a behavioral disturbance is considered extremely unlikely. Modifying the PDC to allow single beam sonar to operate as low as 24 kHz changes means that single beam sonar may operate within the hearing range of low-frequency cetaceans (blue, fin, and sei whales) and mid-frequency cetaceans (sperm whales). However, as discussed in SARBO Section 3.1.8.3, with the exception of North Atlantic right whales, the ESA-listed whales in the area are predominately located off of the continental shelf in waters deeper than where work covered under this Opinion would occur and would not be expected to be present in areas affected by noise from vessels and dredging covered under this Opinion. Geophysical surveys covered under the 2020 SARBO are used in association with these vessels and dredging and therefore single beam surveys are not expected to occur in areas where low-frequency and mid-frequency cetaceans occur. We therefore conclude that behavioral effects to these species would be extremely unlikely to occur. Further, in the unlikely event that these whales encounter noise associated with single beam sonar, it is expected that any exposure would be brief, given the nature of this equipment, and therefore any behavioral effects would be insignificant. SARBO PDC GG.6 limits all geophysical surveys from occurring in areas where North Atlantic right whales may be present, including single beam sonar. Single beam sonar operating at 24 kHz will still operate outside of the hearing range of other ESA-listed species in the action area. Therefore, the proposed change to PDC GG.4 does not change the outcome of the 2020 SARBO effects analysis for behavioral sound effects, as any effects remain extremely unlikely to occur.

Conclusion

For the reasons stated above, we have determined that there is sufficient information to support the proposed change to PDC GG.4. Additionally, the proposed change to PDC GG.4 does not diminish the PDCs' minimization of impacts of takes to ESA-listed species associated with 2020 SARBO, as all anticipated effects associated with PDC GG.4 as modified would remain extremely unlikely to occur. We have also determined that the proposed change to PDC GG.4 does not trigger reinitiation pursuant to 50

C.F.R. § 402.16, as the proposed modification will not result in effects to ESA-listed species or designated critical habitat beyond those considered in the 2020 SARBO.

The project has been assigned a tracking number in our new NMFS Environmental Consultation Organizer (ECO), "SERO-2019-03111." Please refer to the ECO tracking number in any future inquiries regarding this project.

The revised document may be accessed here: https://www.fisheries.noaa.gov/content/endangered-species-act-section-7-biological-opinions-southeast

We look forward to further cooperation with you on other projects to ensure the conservation of our threatened and endangered marine species and designated critical habitat. If you have any questions on this consultation, please contact Karla Reece, Section 7 Team lead, by email at karla.reece@noaa.gov.

Sincerely,

Roy E. Crabtree, Ph.D.
Regional Administrator

Enclosure (s)
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